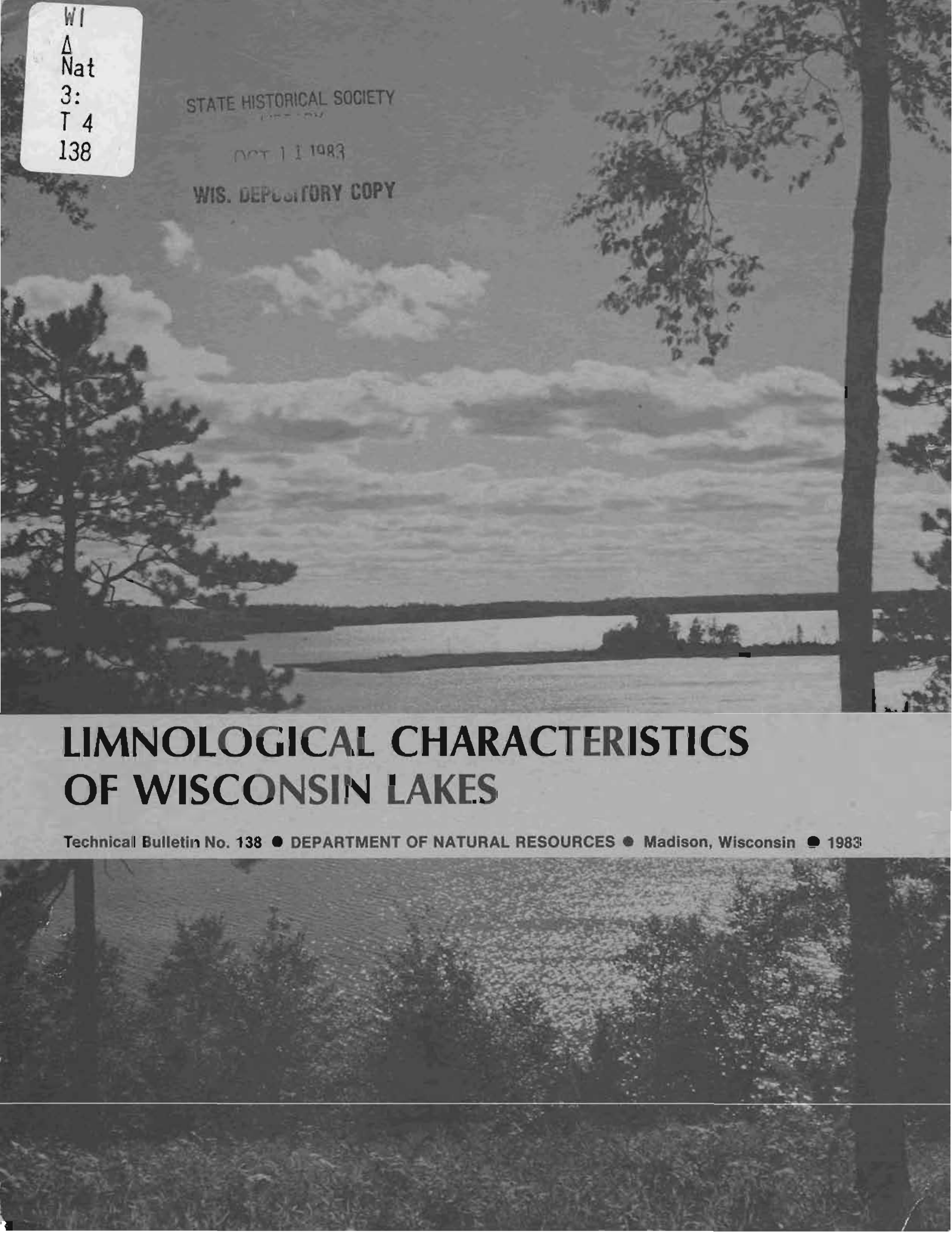

LIMNOLOGICAL CHARACTERISTICS OF WISCONSIN LAKES

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ABSTRACT

This report is the culmination of 14 years of extensive lake data collection conducted primarily by many individuals in the Bureau of Research Water Resources Section.

Data gathered during this time span, supplemented by the data collected in 1979 through a comprehensive random sample of 25% of all Wisconsin lakes and impoundments greater than 5 ft deep and 25 acres in size, allowed the evaluation and examination of general limnological characteristics of Wisconsin lakes.

Descriptions of ranges, medians, and means are provided for common water quality parameters (physical, chemical, and biological) on the basis of statewide and regional distributions and also on the basis of general lake types.

An apparent water quality index for Wisconsin lakes has been developed based on the major parameters by which lake water quality and/or trophic classification are currently judged — water clarity, chlorophyll *a*, and total phosphorus concentrations. The many factors influencing this index and the parameters on which it is based are discussed in detail.

General interrelationships between the many physical, chemical, and biological characteristics of Wisconsin lakes are presented in respect to both the total data set and several subsets based on lake types.

Evaluations of historical trends in lake water quality are made insofar as possible based on this data set and the limited historical data available on Wisconsin lakes.

The significance of the data base and the possible implications for lake management applications and future lake water quality assessment are presented.

LIMNOLOGICAL CHARACTERISTICS OF WISCONSIN LAKES

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SUMMARY

This report is primarily intended to serve as a technical reference for professionals in any of a number of water-related fields. The water quality data presented here therefore are largely of a specifically technical nature. There is, however, considerable information throughout the report which will be of a more practical value to other resource managers and users. To assist in locating particular results or discussions of the data, we have provided detailed contents and summaries for each chapter.

Over a 14-year period, 1,140 lakes throughout Wisconsin were visited periodically (some only once, some seasonally for several years). Water samples collected during these visits were analyzed for physical, chemical, and biological data which together with field notes and known morphometric characteristics formed the core of the data used in this study and subsequent report.

All of the available data for each lake were condensed into values representing seasonal or annual means for each parameter (e.g., summer total phosphorus levels). This information, combined with classification data (e.g., seepage characteristics) and static data (e.g., depth, size, watershed size), was entered into computer storage and subsequently analyzed by a number of strategies (e.g., statewide, regional, lake type, season, etc.) to form the bulk of the data presented.

While the following generalizations may or may not be "common" knowledge, many of them represent the first time such statements have been accompanied by the facts to support them garnered from a large number of Wisconsin lakes, and in many cases, a wide variety of lake types.

OVERVIEW

Lakes are complex, dynamic ecosystems which are dependent upon a combination of external and internal influences. Direct precipitation on the lake's surface, overland surface runoff including channelized flows (streams, rivers, creeks, etc.), and groundwater inflows serve as the source of lake water. As a consequence, the chemical composition together with many physical (e.g., color, temperature) and biological characteristics (e.g., plankton, bacteria) of the inflowing waters are influenced by anything and everything that the water comes in contact with while enroute to the lake. Therefore, the ionic composition and nutrient loading to any particular lake depend on its watershed (surface and subterranean) and climate. Such factors as geologic structure and composition, soils, vegetation, land use, geomorphology (e.g., slope and drainage patterns) and climatic patterns (e.g., precipitation form, amounts, intensities, etc.) undoubtedly have a great impact on the input of externally generated materials to a lake including not only the amounts and composition, but also the timing of these inputs.

Upon reaching the lake proper, these externally derived inputs dilute or are in turn diluted to various degrees by the concentrations of materials and the volume of water already in the lake. For those lakes with no outlets, the nutrients are essentially trapped and such inputs are generally additive in nature (ignoring the output by harvesting or biological export). In the case of impoundments or drainage lakes, net loss or gain of materials is possible and depends on the state of equilibrium between the chemical composition of inflowing waters and lake water.

The response of any individual lake to this influx of nutrients (in terms of such water quality parameters as nutrient concentrations, water clarity, and standing crop of phytoplankton or macrophytes) depends primarily on its physical conditions (size, depth, stratification, basin morphometry), preexisting chemical state (ionic composition and nutrients), and the interaction of its biota with these factors. In many cases, the internal regeneration of nutrients within a particular lake may greatly exceed the externally generated input of materials. A number of complex interactions including the physical mixing and recycling of nutrient-rich sediments, chemical complexing of iron or phosphorus, precipitation of calcium carbonate, and the impact of biota (i.e., macrophytes, bacteria, plankton, fish) upon internal nutrient recycling mechanisms act alone or in concert to affect the internal generation or retention of nutrients. Thus, lakes with currently low annual inputs of nutrients but with high internal recycling rates may appear similar to lakes with currently high external inputs but low internal recycling rates.

As a result of these varied external and internal influences, it is extremely difficult to characterize typical Wisconsin lakes. Lakes which may be quite alike based on some obvious feature such as depth (i.e., deep vs shallow) may be quite dissimilar in another respect (e.g., drainage vs seepage) which may have a much greater impact on existing water quality conditions than the first characteristic. In such cases, comparisons of water quality made between the two classes or groups of lakes are greatly influenced by the proportion of lakes with the more critical characteristic. For example, thermal stratification may have greater impact on water quality than depth alone — thus while deep lakes have generally better water quality than shallow lakes, this distinction in water quality may simply be the result of having a greater percentage of deep lakes stratified than shallow lakes. This principle is one of the most important facts to be garnered from this investigation and report. While lakes can be categorized according to a large number of contrasting features or characteristics, and while certain gross generalizations can be made concerning their water quality, there will always remain a wide variance in the water quality conditions of any particular group of lakes.

Selection of a 25% random sample of all Wisconsin lakes greater than 5 ft deep and greater than 25 acres in size has provided a very adequate data base upon which the best generalizations can be made in regard to Wisconsin's lakes. The random data base best represents the natural distribution of lakes and lake types within the state.

GENERAL CHARACTERISTICS

1. The "typical" Wisconsin lake is not necessarily best described by the means or medians reported in the text. Rather, a more accurate characterization is achieved by describing the conditions (parameter ranges) which exist in the majority of the state's lakes. Under these constraints, the typical Wisconsin lake may be characterized as natural in origin, equally likely to be of seepage or drainage and stratified or mixed in basic lake type, and probably located in the northern half of the state. The lake is probably less than 100 acres in size, less than 30 ft deep, and has a water clarity of less than 3 m (10 ft). Chlorophyll levels are likely less than 10 µg/l, and macrophytes cover only 10% or less of the total lake area. Alkalinity and many ions generally depend upon geographic distribution, with northern lakes more apt to be lower in alkalinity and ionic concentrations than southern lakes. Nutrient concentrations are quite variable and primarily depend upon a number of overlapping influences.

2. Seepage lakes generally have lower concentrations of nutrients and ions and better water clarity than drainage lakes. No significant difference in phytoplankton standing crop as measured by chlorophyll *a* concentration occurs between seepage and drainage lake types.

3. Natural lakes generally have lower concentrations of nutrients and ions and better water clarity than impoundments. There is no significant difference in phytoplankton standing crop as measured by chlorophyll *a* concentration between natural lakes and impoundments.

4. Deep lakes (based on either mean or maximum lake depths) generally have better water quality and lower nutrients than do shallow lakes. No significant differences occur between alkalinity, pH, chloride, calcium, or magnesium.

5. Thermally stratified lakes generally have better water quality and lower nutrients than do mixed lakes. Alkalinity, pH, chloride, calcium, and magnesium concentrations are not significantly different.

6. There is strong evidence to suggest that thermal stratification is more important in determining the response of a lake to nutrient inputs than either mean or maximum depth alone.

7. Lakes which appear "blue" and "clear" generally have better water quality than lakes which

appear "green" or "turbid", and always have chlorophyll a levels less than 15 µg/l (95% have less than 10 µg/l).

8. Nutrient dynamics in lakes are highly dependent upon lake type. Phosphorus decreases from spring to summer in most stratified lakes while it increases in most mixed lakes.

9. Phosphorus appears to be the critical nutrient limiting chlorophyll a standing crop in all but a very few Wisconsin lakes where nitrogen may be limiting.

10. Regional analysis suggests that higher levels of some ions and nutrients exist in southern or southeastern lakes, probably resulting from a combination of existing geological differences and land use patterns. Lakes in the northern regions of the state generally are of better water quality and lower in nutrients although a great deal of these differences may be attributable to the greater number of deeper, stratified, seepage lakes.

11. The mean depth of natural lakes when unknown may be estimated to be roughly half the maximum depth.

12. An equation is provided in the text for estimating a lake's retention time given its drainage basin and lake area dimensions.

13. Deep stratified lakes generally have better water quality and are less likely to experience severe or moderate dissolved oxygen stress than shallow stratified lakes.

14. Lakes experiencing low wintertime dissolved oxygen concentrations generally are richer in nutrients and exhibit poorer summertime water quality than do lakes which do not appear to winterkill.

Water clarity, most often measured by Secchi disc, was one of the indicators used in this study for developing a water quality index for Wisconsin lakes.

WATER QUALITY-TROPHIC STATE

1. An apparent water quality index is presented for Wisconsin lakes, incorporating water clarity, chlorophyll concentration, and total phosphorus. Preferably, all three parameters should be used in evaluating a particular lake.

2. The relationship between water clarity and chlorophyll *a* depends on lake type and is affected by other parameters such as water color and turbidity. Chlorophyll *a* appears to have the greatest impact on water clarity when levels exceed 30 µg/l.

3. Seasonal changes in water clarity depend on lake type and nutrient status. Greater variation was noted in stratified lakes than in shallow mixed lakes.

4. The relationship between total phosphorus and chlorophyll *a* varies with lake type. Mixed lakes generally had a higher concentration of chlorophyll *a* per unit of phosphorus than did stratified lakes.

5. Thirty µg/l of total phosphorus appears as a more reliable predictor of visible chlorophyll *a* levels (10 µg/l or greater) than 20 µg/l total phosphorus.

6. Phosphorus concentrations exhibit a greater relative degree of variability in lakes than do nitrogen concentrations.

7. Total phosphorus concentrations tend to decrease from spring to summer in stratified lakes.

8. Spring concentrations of inorganic nitrogen and inorganic phosphorus in excess of 30 µg/l and 10 µg/l, respectively, tend to produce visible amounts of phytoplankton during the following summer in natural lakes. Lower spring nutrient levels may produce visible concentrations of phytoplankton the following summer in impoundments.

9. A lake's biological community has a great impact on the cycling of nutrients and thus may greatly affect the lake's perceived water quality or trophic status.

INTERRELATIONSHIPS

Strong relationships exist between a great number of measured parameters. Physical characteristics appear to have significant impacts upon many chemical characteristics, e.g., strong inverse relationships between mean depth and nutrients. Many parameters are closely associated, e.g., lakes with high nutrient levels are also high in ionic composition.

On the other hand, a considerable amount of scatter occurs in plots of the various interrelationships, which suggests that caution must be used in applying linear regression formulae derived from these associations for predictive purposes.

HISTORICAL TRENDS

1. Assessment of trends in water quality data on Wisconsin lakes is restricted by an insufficient number and/or frequency of samples and differences in sampling and/or analytical methodologies.

2. There is no conclusive evidence that would indicate any Wisconsin lake has experienced permanent change in pH or alkalinity since early this century.

3. Chlorides have increased significantly in many southern Wisconsin lakes.

4. While increases in nitrogen and phosphorus may be occurring in some Wisconsin lakes, data are insufficient to document nutrient increases other than in those lakes which are known to receive point-source sewage discharges.

5. Although water clarity is one of the most important water quality indicators, it is one that generally shows great variability and we were not able to pinpoint long-term changes.

INTRODUCTION

BACKGROUND

Wisconsin has nearly 15,000 lakes of such great diversity in origin, configuration and chemical and biological composition that they almost defy categorization and classification. Natural or man-made lakes are found in every county in the state, but greatest numbers lie across the north and eastern parts of the state while very few exist in the driftless area of southwestern Wisconsin (Fig. 1). Wisconsin inland lakes range in size from less than an acre to 137,000 acres, and in depth from a few feet to 230 ft. Only about one-third have been named, and almost two-thirds are less than 10 acres in surface area (Fig. 2). Lakes 25 acres in size or larger constitute over 93% of the nearly million acres of lake surface area in the state, although only 20% of the lakes are in this size category.

It is a foregone conclusion that lakes comprise one of the state's most important natural resources and economic assets. They are the basic ingredient for a host of water-based recreational activities and serve as one of the major attractions in the state's multi-billion dollar tourism industry. In addition, these lakes constitute an immeasurably valuable aesthetic resource, and in some cases are very important for specialized uses such as drinking water supply and industrial cooling. Because of these values, it was inevitable that the lakes in Wisconsin would be prime attractors of people, which has been the case historically and still is today. As a result, settlement and development of the shorelines of many Wisconsin lakes have proceeded rapidly, and the use of lakes for recreational pursuits began early and continues to flourish.

HISTORY OF WISCONSIN LAKE STUDIES

Not only the settlers, developers, and recreationists recognized the value of Wisconsin's lakes and were attracted by them, but scientists at the University of Wisconsin also became interested in them at a very early date. Lake studies were initiated by E. A. Birge at the University of Wisconsin (UW) in the late 1800's and were carried forth under the auspices of the Wisconsin Geological and Natural History Survey, established in 1897. In the years that followed, Birge and his co-worker Chauncey Juday conducted studies of Wisconsin lakes that became world renowned. In recognition of their pioneering efforts in the field of limnology, the period of years beginning before the turn of the century and continuing until the 1940's has become known as the "Birge and Juday Era" of limnology. The accomplishments and contributions made by these two men are related in detail by Frey (1963); the data bank they and their associates compiled on Wisconsin lakes has served as an invaluable source of information for later investigators. Further, their remarkable understanding and description of limnological processes were such that in many cases lake studies made in

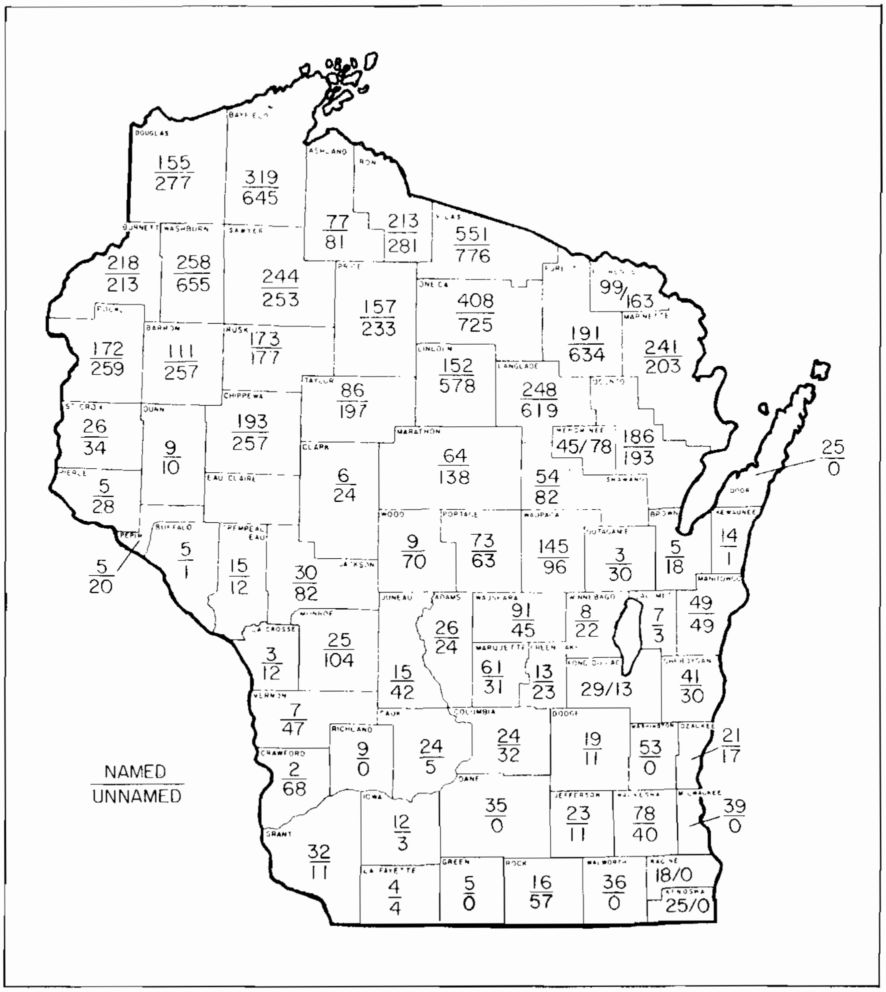


FIGURE 1. *Numbers of named and unnamed lakes in each Wisconsin county.*

later years were unable to provide any new information.

Because of the impetus given to the study of lakes during the Birge and Juday Era, it is not surprising that interest in Wisconsin lakes has remained high among both scientists and citizens of the state. At the University of Wisconsin, the traditions in limnological research established by Birge and Juday were carried forward by their co-workers and protégés. Many different Departments (Zoology, Botany, Geology, Soils, Bacteriology, Water Chemistry, Engineering and others) and branches of the University were involved directly or indirectly in lake research, and a variety of studies dealing with different aspects of the limnology of Wisconsin lakes has been published. However, chemical and biological data were not collected on large numbers of lakes in the state following the Birge and Juday Era, hence a considerable gap in the historical data base on Wisconsin lakes exists between the original Birge and Juday data and data collected in recent years.

The Wisconsin Conservation Department and its successor agency, the Department of Natural Resources (DNR), also developed and have maintained active programs for the study and management of lakes. Countless investigations of Wisconsin lakes have been conducted which have added to the knowledge of the physical, chemical and biological characteristics of the state's lakes. Of special significance was the Surface Water Inventory and Classification program that began in 1959. Through this program, all lakes in the state were inventoried by county, with data collected on physical characteristics, water quality, fish and wildlife resources, recreational use, problems limiting use, and various other subjects. Publications on surface water resources have been prepared for nearly all counties of the state. In addition, lake use reports have been written for some individual lakes, and other miscellaneous publications on lake water quality were prepared as an outgrowth of the inventory program (Poff 1961, 1967, Sather et al. 1970-74).

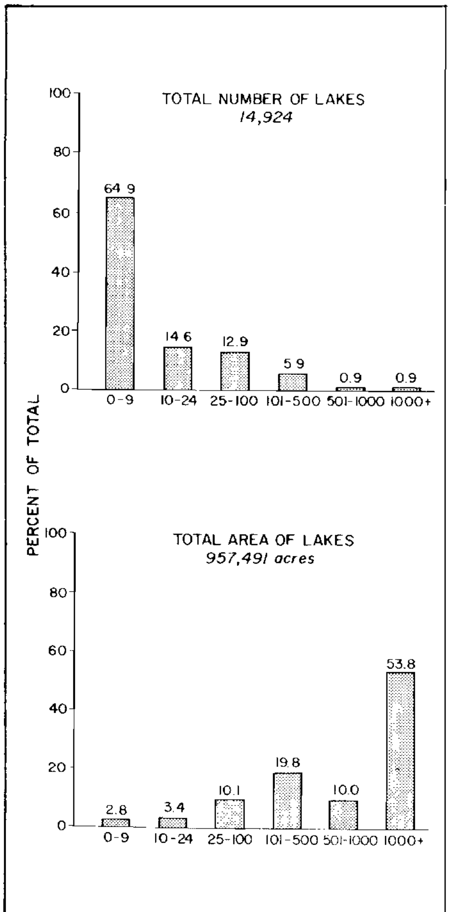


FIGURE 2. *Frequency of lake numbers and total acreage for various size ranges of Wisconsin lakes. (After Lathrop et al. in prep.)*

Other important studies relative to classification and trophic status of Wisconsin lakes have been made by Lueschow et al. (1970), Uttormark and Wall (1975), and the U.S. Environmental Protection Agency (EPA) (1972). Lueschow and co-workers sampled 12 lakes of different type and trophic state and their ranking of these lakes based on various water quality parameters represented the first attempt to develop a trophic status classification system for Wisconsin lakes. Classification of the state's lakes according to trophic status was further explored by Uttormark and Wall when they developed a Lake Condition Index (LCI) for 1,150 lakes, 100 acres in size or greater. They later expanded their index to additional lakes and compared the Lake Condition Index with other trophic indexes of Wisconsin lakes. The EPA studied 46 Wisconsin lakes in 1972-73 as part of the National Eutrophication Survey; a series of Working Papers were published assessing the trophic state of each sampled lake.

Current ongoing programs of lake study are the remote sensing and inland lake renewal programs. Studies to determine trophic status and classify lakes by remote sensing began in 1974 and are continuing through a cooperative project of the Wisconsin DNR, UW, and EPA (Martin and Holmquist 1979). The inland lake renewal program, initiated in 1974 by legislative action, is a major effort to protect and rehabilitate Wisconsin lakes. Through this program, which involves the Wisconsin DNR, UW, EPA, and local units of government, funding has been provided to date for studies of more than 100 lakes in the state to determine the feasibility of various alternatives for protection or renewal.

THE BUREAU OF RESEARCH LAKE STUDY AND THIS REPORT

A 14-year study of Wisconsin lakes conducted by the Water Resources Research Section, Bureau of Research, will be covered here. This study began in 1966 after the Wisconsin Legislature, recognizing the need for a program of research and data collection on the state's water resources, authorized the study through enactment of Chapter 502, Laws of 1965 — the Interagency Water Resources Research and Data Collection Program (6-S). Funding of the 6-S program was not renewed after 1972, but the lake study was continued through funding from other state sources. Data collection activities were terminated in 1979, after a data file had been established with information on 1,140 Wisconsin lakes.

The Bureau of Research lake data file has already been widely used as an information source for many DNR programs, and has likewise found use by many other agencies and individuals, but the data have not been previously integrated or analyzed in depth. For this report the lake data have been extensively analyzed in order to meet the following objectives: (1) document existing water quality conditions in Wisconsin lakes, (2) describe some physical, chemical and biological phenomena which are characteristic of Wisconsin lakes, (3) investigate relationships between various water quality parameters and other factors which affect lake water quality, (4) evaluate changes in some water quality parameters which have occurred or may be occurring, and (5) relate findings to possible management strategies, water quality objectives, and lake rehabilitation or protection efforts.

In the historical overview, only some important Wisconsin lake studies have been mentioned, since these are considered to have the most significance relative to the work covered in this report. The great volume of literature on lakes accumulated elsewhere in the United States and in other parts of the world could not possibly be referenced here. However, we are cognizant of the many limnological advances made elsewhere that are applicable to Wisconsin lakes, and have wherever possible applied current knowledge to data analysis and interpretation. Water quality characteristics and interrelationships for Wisconsin lakes have been examined in relation to the findings of other investigators when meaningful comparisons could be made.

Although much of the information found in this document is believed to be of general interest, the report has been prepared primarily as a technical reference for professionals in the fields of limnology, lake protection and management, fish management, water quality planning and management, and other related fields. Because of the volume and nature of the material presented, an effort was made to group together information of special interest to different potential users. Some recapping and cross-referencing between sections of the report were necessary using this organizational format in order to make each section as nearly a separate entity as possible. To facilitate use of the report, subheadings are liberally used to identify the subject material found in each section.


State Historical Society of Wisconsin

University of Wisconsin researchers E. A. Birge (left) and C. Juday (right) on Lake Mendota in about 1917. Their pioneering work in the limnology of Wisconsin lakes became world renowned.

PROCEDURES

SELECTION OF LAKES

The Bureau of Research lake study program underwent many changes and varied considerably in scope and direction throughout its 14-year history. During this period, several different approaches to lake selection and sampling were taken. Originally, lakes were selected by Bureau of Research personnel on the basis of their importance as a resource, size, public use, aesthetic or scientific value, and potential for degradation. Thus, many of the state's largest and most heavily used lakes were among the first to be sampled. As interest in lake water quality expanded within the DNR, a selection system was initiated that allowed all interested persons within the DNR to nominate lakes for inclusion in the program. District and Bureau staffs were solicited biennially for names of lakes on which they wished to have water quality data collected. Under this selection system, which was in effect until spring 1979, many "problem" lakes were sampled — lakes with weed and/or algae problems, use problems, or other management problems. As a result, the data file on the 561 lakes sampled from 1966 to early 1979 shows a bias toward lakes in the following categories: eutrophic, weedy, algae blooms experienced, impoundments, large size, easily accessible, and located in more heavily populated parts of the state. Although these data provide a set of water quality information on lakes of great importance and interest, it is one that is not a representative cross-section of lake types existing in all parts of Wisconsin.

In order to compensate for the biases in the existing data file and provide a more complete base of information on various Wisconsin lake types, a set of lakes was selected randomly by the Bureau of Research for sampling in the summer of 1979. This was a one-time data collection effort geared to provide information on a representative cross-section of all lake types found statewide. Lakes included in the program were selected randomly by county from a list of lakes 25 acres in size or larger and greater than 5 ft maximum depth. Water bodies less than 5 ft deep were eliminated from the selection process because they were considered to be deep marshes rather than lakes, and river backwaters and artificially flooded cranberry marshes were also eliminated as not being true lakes. A total of 2,802 lakes were on the list for possible selection, and 700 (25%) were selected (Table 1). In order to ensure a geographically diverse sample, the 25% random selection process was carried out on a county-by-county basis. A number of the lakes (86) selected for sampling in 1979 had been previously sampled in 1976-78 and the summer data on these lakes were used in the random sample analyses. Also, 54 of the randomly selected lakes were sampled by the North Central District of the DNR in 1979 as part of their lake sampling program.

The random sampling program fell slightly short of the planned 25% sample — the actual sample was 24% (661). Also, a certain number of substitutions had to be made in the field, because some lakes could not be sampled due to access problems. In these cases (40), a nearby lake was chosen that was similar in area and depth to the lake originally selected.

In addition to the randomly selected lakes, all lakes over 1,000 acres in size which had not been previously sampled were sampled in 1979. The data file, therefore, contains information on the 127 largest lakes in Wisconsin, which have a combined area representing 54% of the total lake surface area in the state.

LAKE SAMPLING

The sampling program as it was carried out through spring 1979 became known as the "quarterly lake sampling program" because lakes were routinely sampled in winter, spring, summer and fall. Attempts were made to collect spring and fall samples at a time when the lakes were completely mixed, although sometimes it was not possible due to short duration of the mixing period. Winter and summer sampling trips were usually made late in the season in order to document the most severe conditions of stagnation. The number of sampling trips made to each lake on the monitoring list varied considerably depending on the need for information: most lakes were sampled each season for two years, but some were only sampled once, and others were sampled quarterly for as long as five years.

The 1979 random sampling program involved sampling of each lake only once during the period of maximum summer stratification in the months of July, August and early September.

Lakes were sampled at the point of maximum depth or at some other location considered to be best representative of the water body. Temperature and dissolved oxygen profiles of the water column were made, and water clarity was measured with a Secchi disc (Table 2). At least two water samples were always collected for laboratory analysis from each lake sampled before summer 1979 — one from near the surface and one from near the bottom. Mid-depth samples were also taken from the lower thermocline when lakes

TABLE 1. *1979 random lake sampling program for lakes 25 acres or larger in size and greater than 5 ft maximum depth.*

Code	County	Total No. in County	25% Random Sample	No. Sampled 1976-78	No. Sampled 1979	No. Substitutes Sampled 1979	No. Sampled by NCD 1979	Total No. Sampled	Percent Sampled
1.	Adams	17	4	1	2			3	18
2.	Ashland	44	11		9	1		10	23
3.	Barron	67	17	1	15	1		17	25
4.	Bayfield	147	37	1	32	3		36	25
5.	Brown	1	0					0	
6.	Buffalo	1	0					0	
7.	Burnett	140	35	2	34			36	26
8.	Calumet	1	0					0	
9.	Chippewa	46	12	2	10			12	26
10.	Clark	6	2	1		1		2	33
11.	Columbia	13	3		3			3	23
12.	Crawford	0	0					0	
13.	Dane	16	4	2	2			4	25
14.	Dodge	9	2		2			2	22
15.	Door	6	2		2			2	33
16.	Douglas	71	18	1	16	1		18	25
17.	Dunn	5	1		2			2	40
18.	Eau Claire	6	2	1	1			2	33
19.	Florence	50	12	3	8	1		12	24
20.	Fond du Lac	10	3		3			3	30
21.	Forest	82	21	1	16	2		19	23
22.	Grant	0	0					0	
23.	Green	3	1	1				1	33
24.	Green Lake	9	2		1	1		2	22
25.	Iowa	5	1	1				1	20
26.	Iron	98	24		19	4		23	24
27.	Jackson	13	3		2			2	15
28.	Jefferson	16	4		4			4	25
29.	Juneau	16	4	1	2			3	19
30.	Kenosha	17	4	4				4	24
31.	Kewaunee	4	1		1			1	25
32.	La Crosse	1	0					0	
33.	Lafayette	1	0					0	
34.	Langlade	59	15	3	9		1	13	22
35.	Lincoln	52	13	1	3	1	6	11	21
36.	Manitowoc	12	3	1	2			3	25
37.	Marathon	35	9	1	4	1		6	17
38.	Marinette	63	16	1	13	2		16	25
39.	Marquette	26	6	1	5			6	23
40.	Menominee	26	6		5	1		6	23
41.	Milwaukee	0	0					0	
42.	Monroe	27	6		5			5	19
43.	Oconto	76	19	4	13			17	22
44.	Oneida	271	68	3	24	1	33	61	23
45.	Outagamie	1	0					0	
46.	Ozaukee	6	2		2			2	33
47.	Pepin	3	1		1			1	33
48.	Pierce	0	0					0	
49.	Polk	107	27	5	19	1		25	23
50.	Portage	34	8	2	6			8	24
51.	Price	75	19		16	3		19	25
52.	Racine	10	3	2	1			3	30
53.	Richland	3	1		1			1	33
54.	Rock	9	2	1		1		2	22
55.	Rusk	34	8	2	5			7	21
56.	St. Croix	19	5		5			5	26
57.	Sauk	12	3	1	2			3	25
58.	Sawyer	120	30	4	23	3		30	5
59.	Shawano	28	7	1	6			7	25
60.	Sheboygan	13	3	1	2			3	23
61.	Taylor	28	7	1	5			6	21
62.	Trempealeau	6	2	2				2	33
63.	Vernon	4	1		1			1	25
64.	Vilas	358	90	13	48	5	14	80	22
65.	Walworth	28	7	7				7	25
66.	Washburn	173	43	1	38	5		44	25
67.	Washington	16	4		4			4	25
68.	Waukesha	45	11	2	7	1	1	10	22
69.	Waupaca	49	12	1	11			12	25
70.	Waushara	37	9	1	7			8	22
71.	Winnebago	5	1		1			1	20
72.	Wood	11	3	1	1			2	18
	TOTAL	2,802	700	86	481	40	54	661	24

were stratified. Chlorophyll *a* measurements represented surface samples only. During the summer 1979 random survey, a composite was made from samples drawn from 0-, 3- and 6-ft depths for laboratory analyses and chlorophyll *a* determinations. In stratified lakes an additional water sample was taken from the middle of the hypolimnion for laboratory analysis. From 1966 through spring 1979, biological samples were collected as time permitted and interest dictated. Information was collected on aquatic macrophytes, phytoplankton and zooplankton in various types of lakes throughout the state. Zooplankton samples were taken during each season of the year from 190 lakes statewide in 1973-74 and later examined by Torke (1979).

Chlorophyll *a* samples were filtered at the end of each collecting day, and the filters were placed in 4 ml of 90% acetone and kept in a freezer for later extraction. Water samples were cooled in ice chests in the field after collection until they could be put into cold storage. The large numbers of water samples generated by the lake sampling program greatly exceeded the laboratory's capacity to analyze them immediately, necessitating cold storage for many of the samples up to three months. However, extensive comparison of these data with lake data collected and analysed by others has shown that sample storage did not seriously affect analytical results (Mason 1980).

LABORATORY ANALYSIS

Nearly all of the lake samples collected during this study were analyzed at the Bureau of Research Laboratory, Delafield, Wisconsin. The few that were not analyzed at Delafield were sent to the State Laboratory of Hygiene or analyzed at the Nevin Laboratory, both in Madison. Analytical results of these laboratories were compared and found to be compatible within acceptable limits (Mason 1980).

Methods used by the Delafield Laboratory in the analysis of lake water samples and their source are shown in Table 2. Since these methods are all standard techniques used by other laboratories and are fully covered in the reference listed, they will not be further discussed. Description of the operational procedures followed at the Delafield Lab and confidence limits for the analyses performed can be found in other publications (Wisconsin Department of Natural Resources 1973-74, 1974-75) and also will not be given here.

TABLE 2. *Analytical methods used in the lake sampling program.*

Parameter	Method	Reference*
Field		
Temperature	Electric thermistor thermometer	(1)
Dissolved oxygen	Modified Winkler	(2)
Transparency	Secchi disc	(1)
Chlorophyll *a*	Filtration and extraction	(5)
Laboratory		
pH	Glass electrode meter	(1)
Alkalinity	Acid/base titration using pH meter	(1)
Conductance	Wheatstone bridge	(3)
Nitrite N	Colorimetric N-(1-napthyl)-ethylenediam	(3)
Nitrate N	Colorimetric brucine sulfate	(3)
Ammonia N	Distillation-Nesslerization	(3)
Organic N	Digestion, distillation, Nesslerization	(4)
Total N	Sum of all nitrogen forms	(3)
Reactive P**	Molybdate colorimetry (unfiltered sample)	(6)
Total P	Acid digestion-molybdate colorimetry	(6)
Chloride	Mercuric nitrate titration	(2)
Metals (calcium, magnesium, sodium, potassium)	Atomic absorption spectroscopy	(4)
Turbidity	Hach model 2100A turbidimeter	(3)
Color	Hellige color comparitor	(3)

*References cited are as follows:

(1) Lind (1974).
(2) American Public Health Association (1975).
(3) Environmental Protection Agency (1974).
(4) U. S. Geological Survey (1970).
(5) Strickland and Parsons (1968).
(6) Eisenreich, Bannerman, and Armstrong (1975).

**Referred to in text as inorganic P.

DATA HANDLING

The Bureau of Research lake data file contains different data sets. For this report these data sets were utilized for different purposes, and were analyzed separately and in combination as follows:

1. The 1966-79 quarterly sampling data (561 lakes): Used for analysis of seasonal variations in lake water quality and for gaining insight into the meaning and significance of seasonal changes. Hereafter in this report the data collected from 1966 through spring 1979 will be referred to as the quarterly data file.

2. The summer 1979 random sample data (661 lakes): Used to describe lake types and characteristics statewide and in different geographic regions, and to determine relationships between various water quality parameters.

3. The large lake data (127 lakes): Analyzed to show the water quality characteristics of the largest (1,000 acres+) lakes and impoundments in the state.

4. The combined (total) data set (1,140 lakes): Used for distributional mapping of various water quality parameters, analyzing historical trends in water quality, assessing management implications and considerations, and describing water quality characteristics and interrelationships wherever the bias inherent in the data set did not preclude its use.

The amount of raw lake water quality data available made it necessary to condense and code some of the data in order to expedite analysis. This section of the report will clarify these manipulations and explain the use of codes and key terminology.

In order to make comparisons between and within certain geographic areas of the state, some basis for aggregation and/or separation was required. Based on a consensus of several experienced limnologists with knowledge of Wisconsin lakes, five lake regions were delineated (Fig. 3). The selection of these lake region boundaries was chosen to: (1) group lakes of similar nature or apparent characteristics, (2) provide sufficient numbers of lakes of various lake types in each region to provide for adequate statistical analysis, and (3) separate lakes on the basis of regional means.

The five lake regions selected resemble the "hydro-chemical lake regions" described by Poff (1961), which were based on bedrock and glacial geology and generalized soil cover typology.

Three stratification classifications were used to code the lakes. Thermal stratification of each lake during the summer period was evaluated for the summer sampling dates. The number of summer sampling dates for each lake varied from one to several, but usually the stratification conditions remained the same from one year to another. If a lake was stratified during one sampling period but mixed or weakly stratified at another, careful examination of pertinent data was made to estimate what classification best represented the summer thermal stratification condition. Lakes distinctly stratified during each summer sampling period were labeled "stratified" while lakes that were not thermally stratified were labeled "mixed". If the stratification status of the lake was questionable (weakly stratified at best), or if the data were inadequate for determining its status, the lake was labeled as such. For most analyses involving stratification, only those lakes clearly "mixed" or "stratified" were used. In a few cases lakes classified as "questionable" were included in the "mixed" category.

The designation of "impoundment" vs "natural lake" is based on the origin of the lake. An "impoundment" refers to a lake created by man through whatever action to impound or impede the flow of a stream or river. A "natural lake" refers to lakes formed by natural events. In Wisconsin this is practically synonymous with glaciation, as the large majority of lakes here were formed or created during the glaciation periods. In all areas except the driftless region of southwestern Wisconsin, the glaciers had direct or indirect influence upon the formation of lake basins or upon the character of their watersheds. For this analysis, a lake formed naturally with a controlled water level was

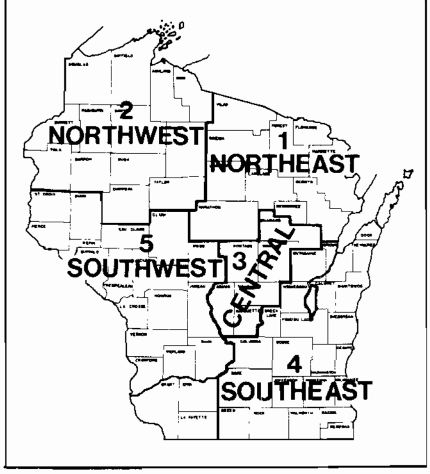


FIGURE 3. *Generalized Wisconsin lake regions.*

classified as a "natural lake". The use of the term "lakes only" throughout the remainder of this report refers to all "natural lakes" exclusive of impoundments.

Separation of "seepage" vs "drainage" refers to the presence or absence of an outlet flow of water. This information was obtained from the county surface water resources publications (Wisconsin Department of Natural Resources 1960-80) or through examination of topographic maps to determine the presence or absence of an outlet flow.

Available dissolved oxygen (D.O.) data consisted of profiles taken at various selected depths to within 1 ft of the lake bottom. Sample sites were usually at or near the point of maximum lake depth but not always. Due to differences in sample collection or depth selection methods, particularly in relation to the selection of sampling depths within and close to the thermocline, it was sometimes impossible to adequately assess the D.O. status of the hypolimnion. Because of this problem, and the fact that sampling dates varied from early to late summer, it was necessary to report only general D.O. conditions. In the system used, lakes with less than 1 mg/l D.O. present in the entire hypolimnion were labeled "severe" D.O. depletion; lakes with D.O. less than 1 mg/l at some position in the hypolimnion were rated "moderate" depletion; and lakes with D.O.'s greater than 1 mg/l throughout were rated "no problem". If more than one set of summer data was available, the most severe conditions observed were used to rate the lake.

Secchi disc or water clarity readings reported may represent either seasonal means or individual discrete values. Secchi disc data used for comparisons with other water quality parameters consisted entirely of single discrete values taken at the same time as the chlorophyll *a* samples, but in the case of the quarterly data set, the other water quality parameters represent seasonal or annual means. This is an important distinction that will be stressed again later. Also of importance is an inherent bias that may exist within the Secchi disc data. Because discs that were visible on the lake bottom or that disappeared in beds of aquatic vegetation were coded as missing data or were included as such in the computation of seasonal means, some of the best water clarity readings were lost from the data base. Although the majority of such cases occurred in shallow lakes, it is possible that this bias may have obscured some details of the analysis, particularly in reference to some of the seasonal analyses.

In the quarterly data set all water quality parameters except chlorophyll *a* represent annual or seasonal means of epilimnetic water samples. The number of samples varied from only one in a few cases to as many as 20 or more in others. Epilimnetic data were chosen over other possible combinations of samples including metalimnion and hypolimnion because they best represented or reflected the water quality as observed by the public. Further, early in the analysis epilimnetic and metalimnetic data were combined to determine if the resulting number would better represent the total in-lake value at the time of sampling. However, this method did not prove to be feasible, since a representative metalimnetic sample sometimes was not collected. The "middle" sample collected often was from below the bottom of the metalimnion and thus represented the upper hypolimnion. At this depth, lakes were quite often anoxic with extremely high nutrient concentrations, which, when combined with the lower values for the epilimnion, resulted in inordinately high values. Also, the metalimnion or hypolimnion quite often represents a very small portion of the entire lake volume. For these reasons, we decided that the epilimnetic data alone best represented the water quality of sampled lakes.

The summer random sample data represent single date, matching data points, which permit accurate correlation-association analysis of various parameters.

COMPUTER ANALYSIS AND DATA STORAGE

All data were transcribed and entered via a remote computer terminal into the University of Wisconsin-Madison Academic Computing Center's Sperry Univac 1100/80 using NEAT, an interactive text editor-processor.

Data analysis was performed using MINITAB, a statistical computing system designed by Ryan, Joiner, and Ryan (1976) (copyright 1980 - Penn State University 1980).

Raw data are on file at the DNR Bureau of Research, Water Resources Research Section, 3911 Fish Hatchery Road, Madison, Wisconsin 53711. Access to the computerized file may be made available to the public. Further information may be obtained by contacting the above address.

ERRATA DHEET

(Please make the following changes to this copy of Technical Bulletin # 138.)

Page 5 Under Water Quality-Trophic State # 8:

Change 30 ug/l to 300 ug/l.

Page 6 In Figure # 1; number of named lakes in Rusk County:

Change 173 to 73.

Page 47 In first paragraph: sentence should read "Another factor to consider is that the presence of toxic materials, high inorganic turbidities, or humic color may inhibit maximum algal production, ..."

Page 77 Formula in 3rd paragraph: change FR to $\sqrt{FR}$ and

Table 37: change FR to $\sqrt{FR}$.

Page 103 In 2nd line of 3rd paragraph: change (> 10 ug/l), 65% to 35%.

Page 111 In "Northeast" matrix: parameter headings should be the same as shown for other matrixes.

Wisconsin's deepest lake, Big Green, in Green Lake County.

GENERAL CHARACTERISTICS

Approximately half of the state's lakes over 25 acres in size are landlocked seepage lakes while the other half are drainage lakes with outlets.

Impoundments make up 16% of the state's lakes over 25 acres in size and have several distinct water quality characteristics.

About 40% of Wisconsin's lakes have brown color of some degree, usually due to humic stain derived from bog surroundings.

INTRODUCTION

The physical, chemical and biological qualities of Wisconsin's lakes and impoundments are highly variable and are influenced by many factors, both external and internal to the lakes. Watershed, or drainage basin, features such as climate, geology, topography, soils, vegetation, and man's cultural activities (i.e., land use) play important roles in determining the quantity and quality of the water entering a lake, while internal factors such as lake basin morphometry and composition and complex biological interactions account for many of the differences in water quality in lakes with similar watershed features.

These factors, together with the widespread geographical distribution of Wisconsin lakes (Fig. 4), greatly limit the extent to which generalizations can be made. However, some patterns exist and these will be briefly discussed as they apply to all lakes in Wisconsin.

STATEWIDE DESCRIPTIONS OF ALL LAKES

The typical Wisconsin lake (greater than 25 acres in size and more than 5 ft maximum depth) may be best described by the means and medians derived from the random survey data (Table 3). These values represent summer water quality conditions which are considered to be very important because summer is the season of maximum lake usage and summertime conditions most strongly influence the public perception of lake water quality. Furthermore, summer water quality generally has been the primary basis for categorizing and assessing the trophic status of lakes. Since some physical and chemical properties of lake waters do not change appreciably throughout the year in certain types of lakes, summer values are also representative of year-round conditions. Seasonal variations in water quality characteristics of Wisconsin lakes and their significance are described in a later section.

Ranges of values for different characteristics of Wisconsin lakes (Table 4) were determined from the total data set collected over the 14-year history of the lake sampling program. Lakes are categorized according to various important physical, chemical and biological traits in Table 5 and Fig. 5. Most of this information will not be discussed in detail but is provided for reference purposes.

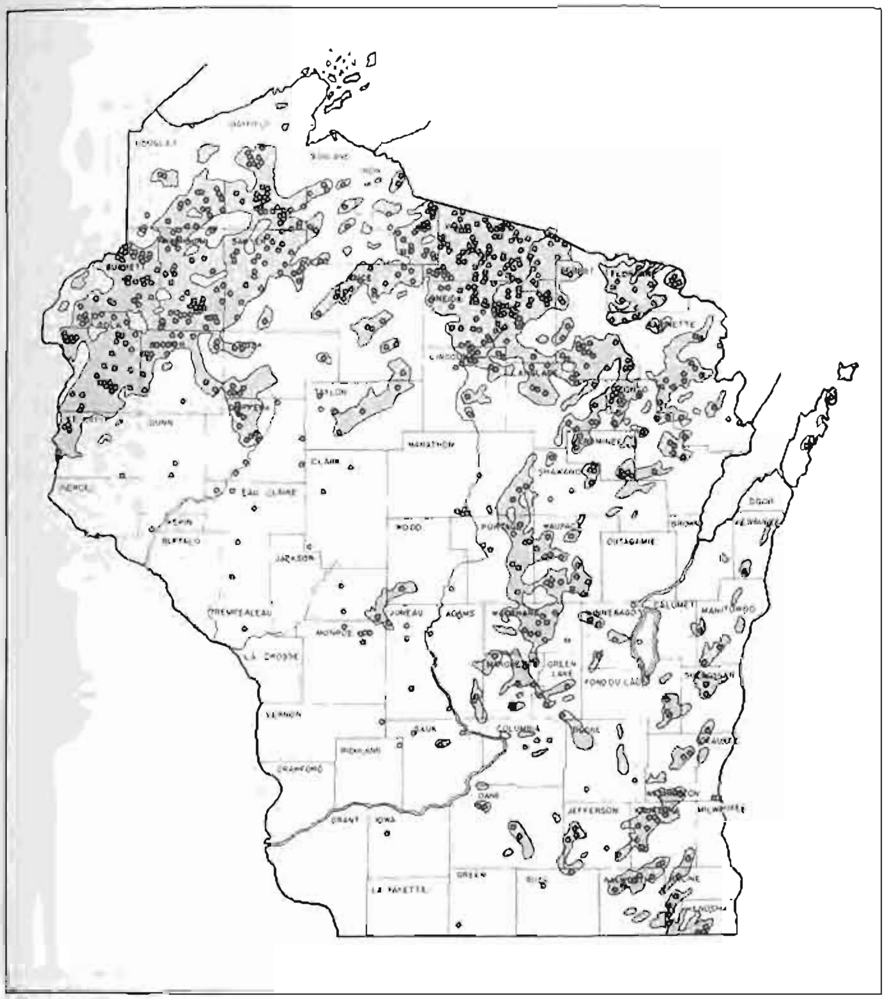


FIGURE 4. *Distribution of 661 randomly sampled lakes (>25 acres and >5 ft maximum depth) shown in respect to the generalized concentrations of the majority of naturally occurring lakes (shaded areas). Most dots outside the shaded areas represent impoundments.*

TABLE 3. *Statistical summary of major characteristics of random data set.*

Parameter	Value	No. Lakes	Mean	Standard Deviation	Minimum	Maximum	Median
Area	acres	660	247	1,104	25	22,218	73
Mean depth	ft	314*	12.2	7.9	2	40	10
Maximum depth	ft	659	25	18	5	105	20
Color	units	560	39	40	1	320	25
Transparency depth	m	595	2.3	1.4	0.1	9.5	2.0
Chlorophyll *a*	µg/l	643	14.8	39.1	0.5	706.1	7.5
Chlorides	mg/l	606	4	7	1	57	2
Calcium	mg/l	604	12	13	1	71	8
Magnesium	mg/l	604	8	11	1	49	2
pH	units	660	7.15	0.85	4.3	9.6	7.2
Alkalinity	mg/l	660	52	59	1	290	30
Turbidity	JTUs	645	3.1	4.6	0.5	72.0	2.1
Organic N	mg/l	659	0.60	0.36	0.10	2.77	0.53
Total N	mg/l	659	0.86	0.57	0.14	6.50	0.73
Inorganic P	mg/l	658	0.013	0.036	0.001	0.570	0.004
Total P	mg/l	659	0.031	0.051	0.003	0.720	0.019

*Only 314 lakes in the random data set had hydrological maps available.

Physical Features

Lake Type

Lake origin and drainage type are important major features affecting water quality. The majority of lakes in Wisconsin (85%) are of natural origin rather than man-made. Their distribution is closely associated with glacial activity. The lakes are evenly divided between seepage and drainage types and with regard to summer thermal stratification (stratified and mixed). Detailed descriptions and comparisons of the characteristics of these different lake types will be presented later.

Area

Lake area or size plays an important role in conjunction with a number of other physical factors in a lake's energy and nutrient budgets. However, there is an overall general lack of correlation between lake area and most water quality parameters (see Interrelationships, p. 79). The average size of the randomly selected lakes was 247 acres with a median size of 73 acres, while the average size of all Wisconsin lakes is known to be less than 10 acres. Approximately 61% of the sampled lakes were in the 25-100 acre range with only about 15% exceeding 250 acres in size. The largest Wisconsin lake is 137,708-acre Lake Winnebago, a shallow lake created by the natural damming of the Fox River in east central Wisconsin. No apparent pattern can be found in the distribution of large lakes within the state (Fig. 6).

Depth

It is very difficult to differentiate between the significance of mean and maximum lake depth since the two parameters are highly associated (Append. A). The establishment of a permanent thermal stratification in summer is dependent upon a lake's maximum depth, among other factors, while mean depth together with lake area determine lake volume and thus the resulting nutrient and ion dilution capacity. The importance of mean depth in influencing lake water quality is well recognized (Rawson 1952 and 1955, Sakamoto 1966, Vollenweider 1968, Ryder et al. 1974, and Schneider 1975).

The average mean depth of the 229 randomly sampled lakes with computed mean depths was 12.2 ft (median = 10 ft). Seventy-four percent of the lakes had mean depths between 5 and 20 ft. Big Green Lake, a natural lake formed by damming of a preglacial valley with morainic materials (Hutchinson 1975), has the greatest mean

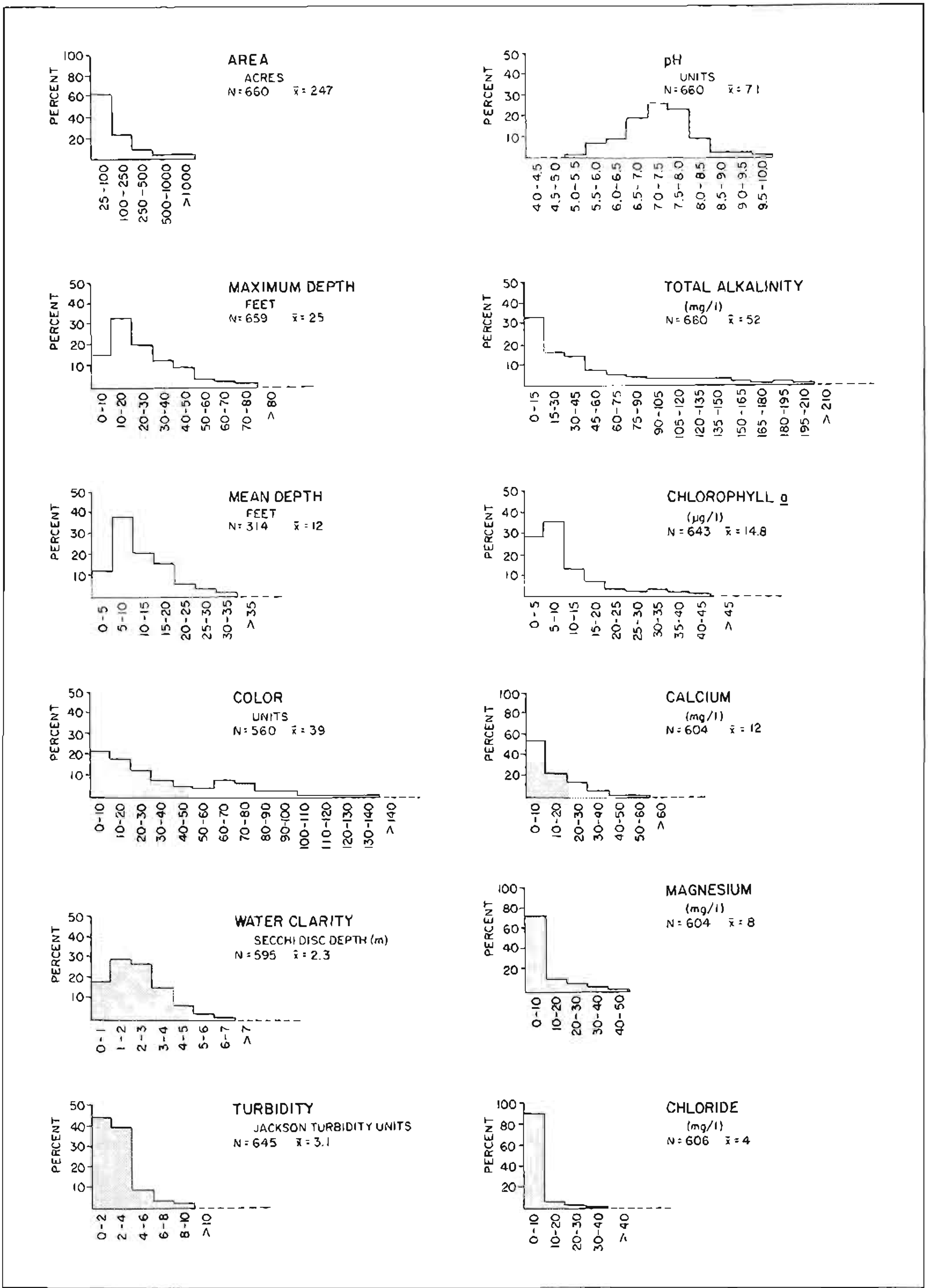


FIGURE 5. *Selected limnological characteristics of Wisconsin lakes within the random data set. (More detailed data may be found in Figure 24.)*

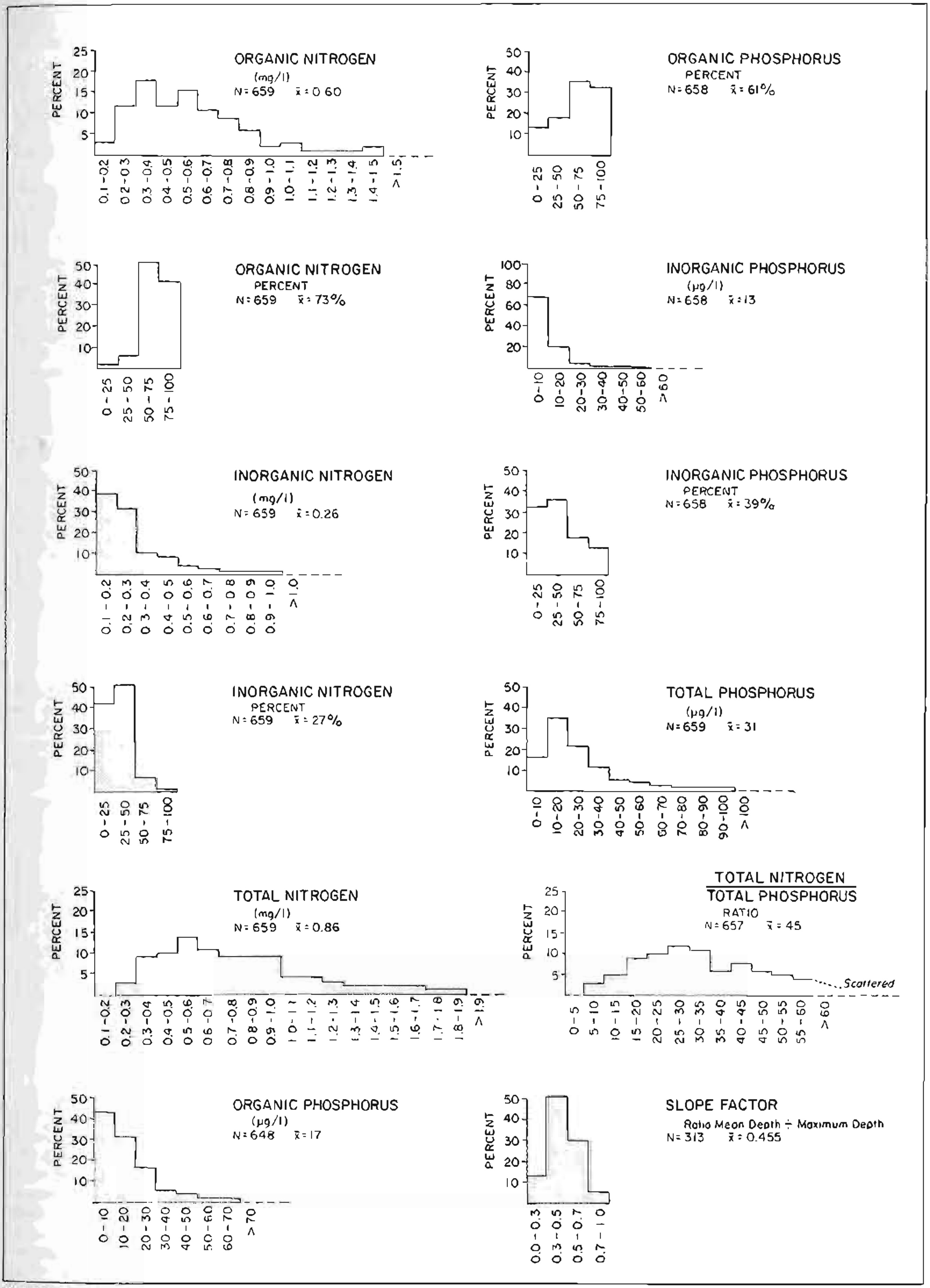


FIGURE 5 (Cont.)

TABLE 4. *Range of values for various characteristics of all lakes sampled during 14-year study period (total data set).*

Parameter		Range (min.-max.)	Measurement	No. Lakes
Area		3 - 137,708	acres	1,302
Mean depth		1.1 - 104	ft	807
Maximum depth		2 - 236	ft	1,309
Color		<1 - 320	units	638
Transparency (summer)		0.1 - 9.6	m	1,192
Chorophyll *a*		<0.5 - 833	μg/l	729
Chlorides		<1 - 269	mg/l	1,252
Calcium		<1 - 90	mg/l	1,250
Magnesium		<1 - 58	mg/l	1,250
pH		4.3 - 9.6	pH units	1,309
Alkalinity		1 - 317	mg/l	1,309
Turbidity		0.4 - 72	JTUs	1,217
Organic N	(w)*	0.06 - 4.25	mg/l	495
	(sp)	0.10 - 2.19	mg/l	507
	(s)	0.02 - 3.21	mg/l	1,287
	(f)	0.03 - 3.58	mg/l	513
Total N	(w)	0.18 - 6.55	mg/l	495
	(sp)	0.12 - 5.06	mg/l	507
	(s)	0.05 - 8.46	mg/l	1,287
	(f)	0.08 - 5.52	mg/l	513
Inorganic P	(w)	0.003 - 0.828	mg/l	494
	(sp)	0.003 - 0.410	mg/l	507
	(s)	<0.001 - 0.736	mg/l	1,285
	(f)	<0.001 - 0.445	mg/l	513
Total P	(w)	<0.009 - 1.074	mg/l	495
	(sp)	<0.009 - 0.70	mg/l	507
	(s)	<0.003 - 1.02	mg/l	1,286
	(f)	<0.009 - 0.57	mg/l	513

* w = winter; sp = spring; s = summer; f = fall.

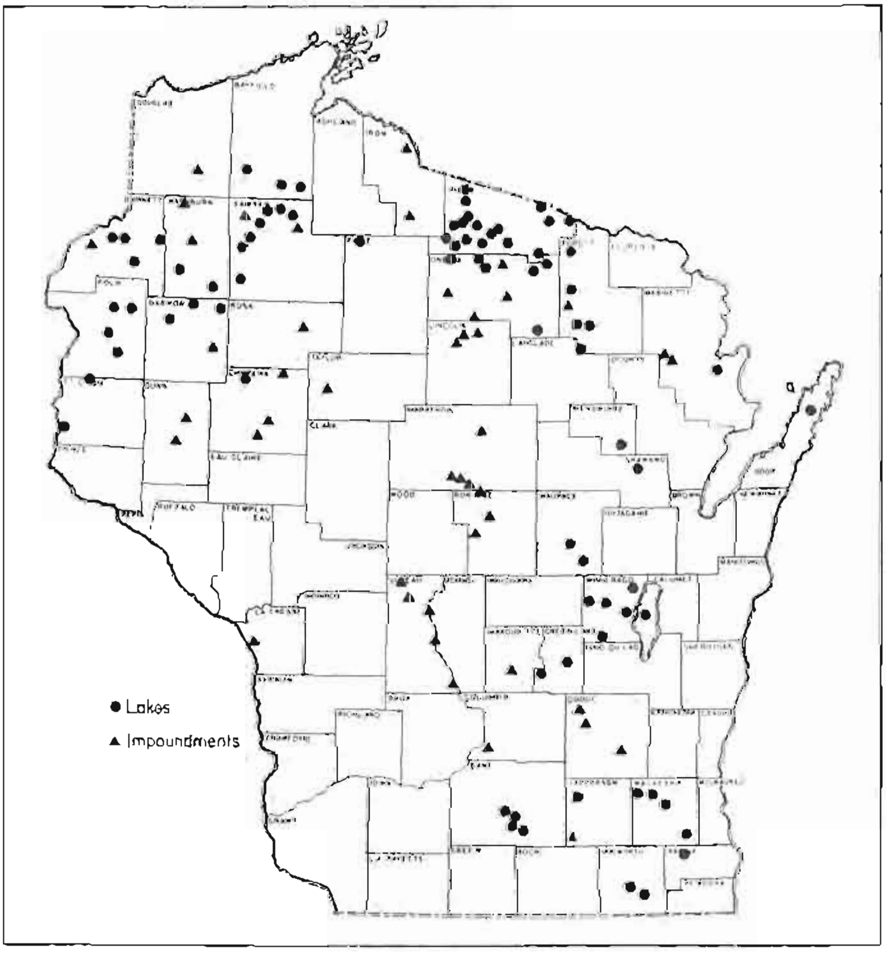


FIGURE 6. *Location of inland lakes and impoundments greater than 1,000 acres in size.*

depth (104 ft) and maximum depth (236 ft) of any inland lake in Wisconsin.

The average maximum depth of the surveyed lakes was 25 ft (median = 20 ft) with a range from 5 to 236 ft. Most lakes (68%) were less than 30 ft deep (maximum depth) with only 10% deeper than 50 ft.

The ratio of mean depth to maximum depth is often related to the shape of a cone in which the ratio of mean to maximum depths would equal 0.33. While the ratio in lakes varies considerably with the shape of the lake, the majority (81%) of Wisconsin's lakes with both known mean and maximum depths lie within the range of 0.30-0.70. The average ratio for Wisconsin lakes — 0.455 (which represents selected lakes and impoundments) — is very similar to the 0.464 value corresponding to an elliptic sinusoid (Wetzel 1975) and the 0.467 reported by Neumann (1959) for 107 worldwide lakes. The same calculation for only the natural lakes in the random data set provides a value of 0.496. This value may be used to estimate a lake's mean depth given its maximum depth; in the case of Wisconsin's natural lakes, mean depth may be simply estimated to be half the lake's maximum depth.

Thermal Stratification

The significance of stratification may be presently underestimated (see discussion under Nutrient Concentrations, p. 71). Stratification appears to be more important than depth alone in affecting summertime lake water quality. Approximately equal percentages of lakes were either mixed or stratified (Table 5). The impact of large lake area or size and the resulting fetch on stratification of Wisconsin lakes is discussed in detail by Lathrop and Lillie (1980). An extreme example of thermal stratification was observed in Third Lake, Trempealeau County, where a 19 C (34 F) change in water temperature was recorded within an 8-ft depth gradient (DNR Bureau of Research Files, 22 July 1975).

Watershed to Lake Area Ratio and Retention Time

The ratio of the size of a lake's watershed to its surface acreage (drainage basin:lake area = DB:LA) and the volume of a given lake are important factors in determining the lake's retention time (RT). A lake with a large DB:LA would theoretically be expected to have a greater volume of surface runoff and/or groundwater influencing it than a lake of similar surface acreage but smaller drainage basin. The lake's RT is considered to be the length of time required for the lake to undergo a complete exchange of water. Two lakes of similar size with equal size drainage basins might be expected to have similar volumes entering them. Under these circumstances, the lake with the greater mean depth, and thus the greater volume, would have a longer RT. These factors have an important function in nutrient dynamics within the lake system and vary considerably with lake type. The combined mean DB:LA for the random data set was 110 and the RT averaged 1.35 years (16.2 months). The median RT for lakes in the random data set was 0.89 years (10.7 months).

Color

Color is an important characteristic affecting water transparency, and heat absorbance and transmission in lakes. The random sample includes data on both measured and perceived water color as recorded during field observations. Many factors can affect perceived color, such as concentrations of dissolved and suspended materials, depth of the water, weather and light conditions, and the angle of observation. Actual or measured water color is primarily dependent on the amount of dissolved substances present; humic materials that contribute various concentrations of organic acids to lake water usually are the cause of the measured brown color found in Wisconsin lakes.

Mean measured water color was 39 units (median = 25); however, color was quite variable (Fig. 5) and showed no clear geographical gradient (Fig. 7). Thirty-nine percent of the lakes had color levels greater than 40 units (Fig. 8). The Rice River Flowage, Oneida County, had the highest color recorded (320 units), which was close to the maximum of 340 units observed during a survey of 518 northeastern Wisconsin lakes by Juday and Birge (1933). A large number of lakes in the southern areas of the state with high measured water color had a yellowish-green color rather than brown, which may have been associated with phytoplankton or related decomposition byproducts.

Thirty-eight percent of the lakes in the random data set were perceived by field crews to some degree as "brown" (Table 5), while 32% were "clear" or "blue", and 20% were "green". Most "brown" lakes were located in the northern half of the state, as shown in Fig. 9.

TABLE 5. *Categorization of Wisconsin lakes by some important types and characteristics (random data set).*

Type or Characteristic	No. Lakes	Percent
Origin or Type*		
Impoundments	100	15
Natural lakes	558*	84
Stratification		
Stratified	283	43
Questionable	40	6
Mixed	261	39
Unidentified	77	12
Drainage Type		
Seepage	332	50
Drainage	329	50
Regions		
Northeast	243	37
Northwest	283	43
Central	44	7
Southeast	61	9
Southwest	30	4
% Macrophyte Cover		
<10%	388	70
10-25%	96	17
25-50%	34	6
50-75%	26	5
>75%	9	2
pH		
<5.0	7	1
5.0-5.9	53	8
6.0-6.9	183	28
7.0-7.9	322	49
>8.0	95	14
Total Alkalinity		
<15 mg/l	227	34
>15<30 mg/l	106	16
>30<90 mg/l	183	28
>90 mg/l	144	22
Summer hypolimnetic dissolved oxygen		
No stress indicated	351	60
Moderate stress indicated	94	16
Severe stress indicated	141	24
Apparent Color		
Green	113	20
Brown	211	38
Turbid	21	4
Clear or blue	177	32
Green and brown	9	2
Brown and turbid	18	3
Green and turbid	4	1

* Three lakes were considered sloughs and were not included.

Water Clarity or Transparency

Good water clarity is a most important recreational and aesthetic attribute of lakes. Only 27% of the lakes randomly selected had water clarity in excess of 3 m in depth when sampled (Fig. 5). Mean water clarity was 2.3 m (median = 2.0 m) (Table 3). A maximum summer water transparency measurement of 9.6 m was recorded on Sparkling Lake, Vilas County, on 27 July 1976 (Table 4). The greatest Secchi disc reading ever observed during the 14-year sampling program was 13.7 m from Thunder Lake, Marinette County, on 25 February 1976. Water clarity showed no well-defined geographical pattern other than a general trend towards better transparency in the northern lakes.

Turbidity

Turbidity is another factor which affects water transparency. Levels of turbidity between 0 and 2 Jackson Turbidity Units (JTUs) were recorded in 44% of the lakes in the random data set, while only 17% of the measurements exceeded 4 JTUs (Fig. 5). The mean turbidity of the randomly selected lakes was 3.1 JTUs (2.1 JTUs median)(Table 3). Highest turbidities were almost always associated with impoundments or shallow lakes where high levels of suspended materials were present.

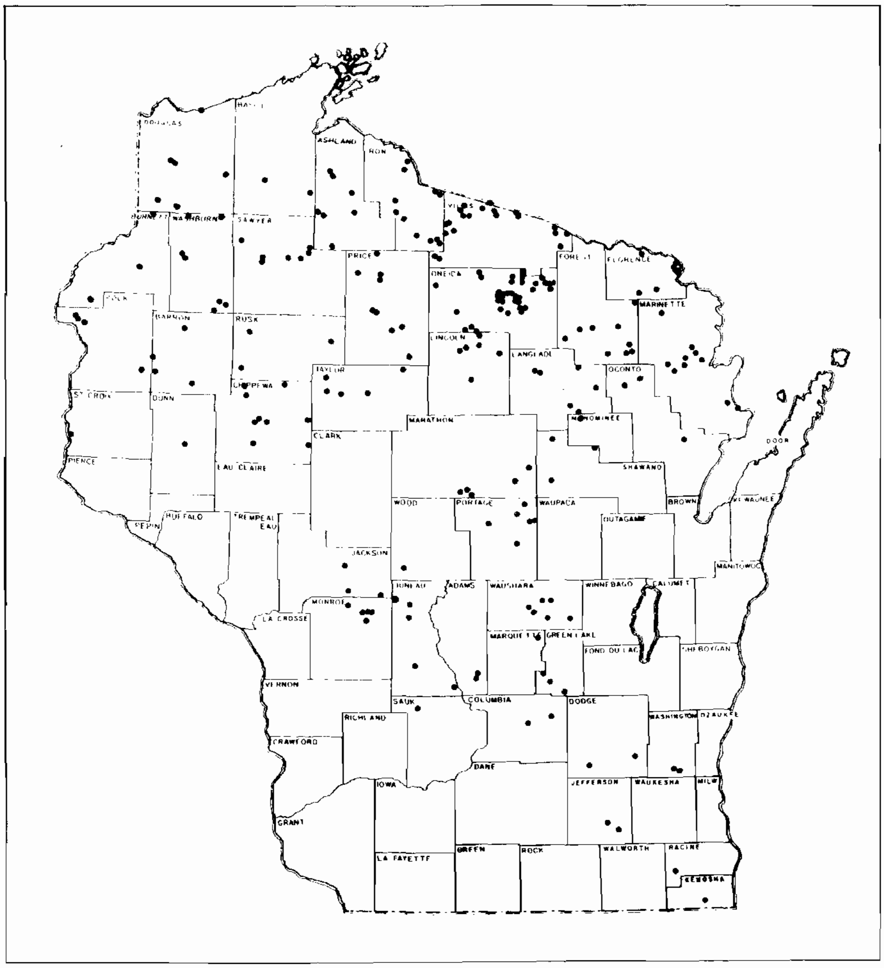


FIGURE 7. *Distribution of Wisconsin lakes with high measured color (>40 units) (random data set).*

Chemical and Biological Features

Chlorophyll *a* Pigment Concentration

The chlorophyll *a* pigment is widely used as an estimation of phytoplankton biomass, and as such it is a very useful parameter by which to compare lakes and lake types. Lakes which appear clear or blue to the eye generally have chlorophyll *a* levels less than 10 µg/l (see later discussion of color, p. 29).

While the reported mean chlorophyll *a* concentration for the random lakes was 14.8 µg/l and the median 7.5 µg/l (Table 3), 65% of the lakes had chlorophyll *a* levels less than 10 µg/l and only 9% had levels greater than 30 µg/l (Fig. 5). The highest chlorophyll *a* concentration recorded during the entire 14-year study — 833 µg/l — accompanied a massive algae bloom on Lake Sinnissippi, Dodge County, a shallow and fertile body of water created by an impoundment of the Rock River (Table 4). No definite geo-

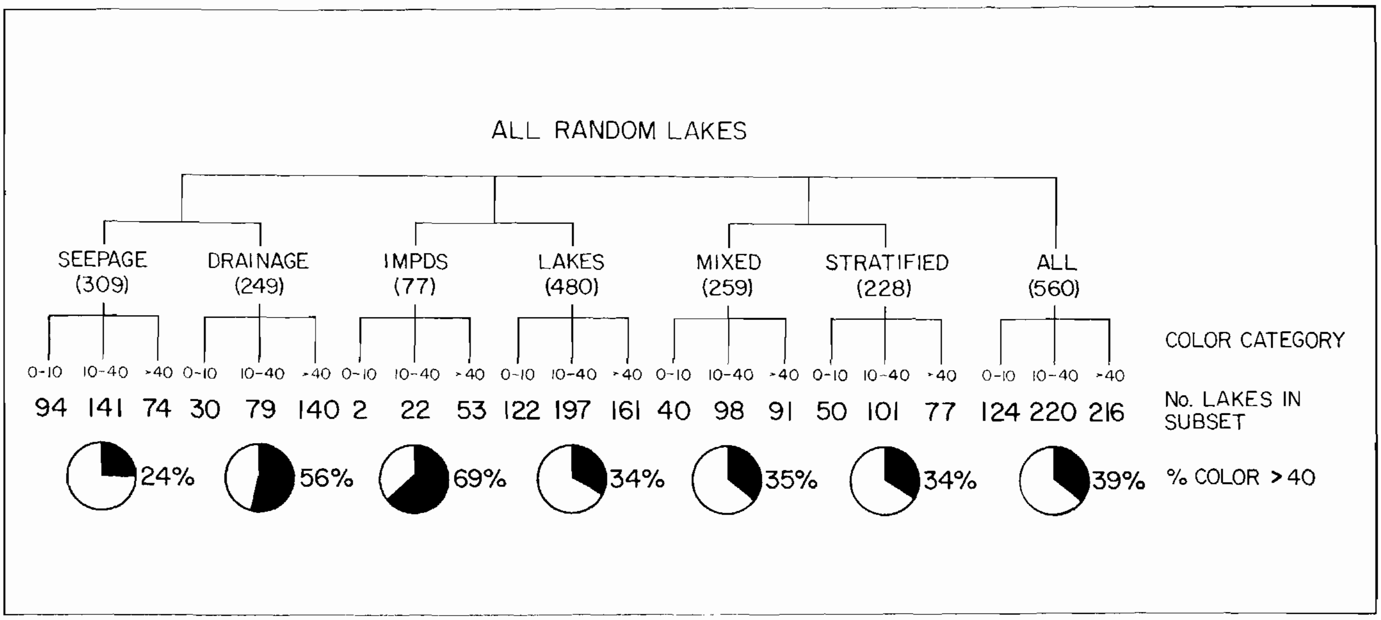


FIGURE 8. *Comparison of numbers of lakes in various color ranges by lake type.*

graphical patterns exist for chlorophyll *a*, but higher proportions of the lakes in the southern areas of the state have high levels.

Macrophytes

Aquatic macrophytes can greatly influence lake water quality; interrelationships between plants and other biological and chemical lake characteristics have been well documented (Wetzel 1975, Carignan and Kalff 1980). Many factors such as morphometry of the lake, light penetration, and nutrient availability affect the extent of macrophyte growth, species composition, and density. Dense growths of macrophytes in strategic areas of a lake's surface can hamper recreational activity and influence certain predator-prey interactions. On the other hand, moderate growths of macrophytes are considered to be beneficial to the well-being of the aquatic ecosystem. Lakes in the random data set were classified by field sampling teams according to the percent of the total lake surface with macrophytes present. Seventy percent of the lakes had little (10% or less) macrophyte coverage, while over 12% had macrophytes covering more than 25% of the total lake surface area (Table 5). Only 7% of all sampled lakes could be considered totally macrophyte dominated (greater than 50% macrophyte coverage). These values do not necessarily represent discrete boundaries corresponding to nuisance conditions since dense stands of submersed macrophytes or marginal beds of vegetation in relatively small areas sometimes create localized lake use problems. Further, observations in the study were made only once during the summer and species composition and density of macrophyte beds often can vary considerably over the course of the growing season.

Alkalinity

Alkalinity, reported as mg/l $CaCO_3$ equivalents, is an index of a lake's buffering capacity (capacity to absorb and neutralize acidic loadings). The alkalinity measurement is primarily dependent on the levels of bicarbonate, carbonate, and hydroxide ions present, which for most Wisconsin lakes reflect the soils and bedrock of the watershed. One half of the lakes in the random survey had alkalinities less than 30 mg/l, while 34% had alkalinities less than 15 mg/l (Table 5). The mean alkalinity (52 mg/l) for the lakes in the random survey was very similar to the mean alkalinity for Precambrian Shield lakes of Canada (Ryder 1964), but considerably lower than that reported by Poff (1970) for another set of Wisconsin lakes. The inclusion of a high number of low alkalinity northern Wisconsin lakes in the random survey accounts for the relatively lower (and more accurate) statewide mean. Hooper (1956) reported a mean alkalinity of 132 mg/l for 241 southern Michigan lakes, while Moyle (1954) reported total alkalinities for various regions of Minnesota ranging from 33 to 201 mg/l. These midwestern values are relatively high compared to the 11 mg/l mean reported for a set of Maine lakes (Davis et al. 1978). Collins Marsh, Manitowoc County, had the highest recorded alkalinity in this study (317 mg/l) (Table 4).

Low alkalinity lakes are mostly confined to the northern half of the state (Fig. 10). The alkalinity map was designed to show where low and moderately low alkalinity lakes are most likely to be found. It differs somewhat from the other distributional maps presented in this report in that the outlined areas contain some lakes with alkalinity concentrations exceeding the value of the isopleth shown. Past events in Wisconsin's complex geologic and glacial history caused soil depths and composition to vary considerably within relatively short distances. These differences, together with numerous climatic and cultural influences, contribute to the rather diverse distribution of relatively low and high alkalinity lakes within given geographical areas. While alkalinity gradients are not apparent, it may be safely stated that the highest alkalinity lakes occur in the southeastern portion of the state.

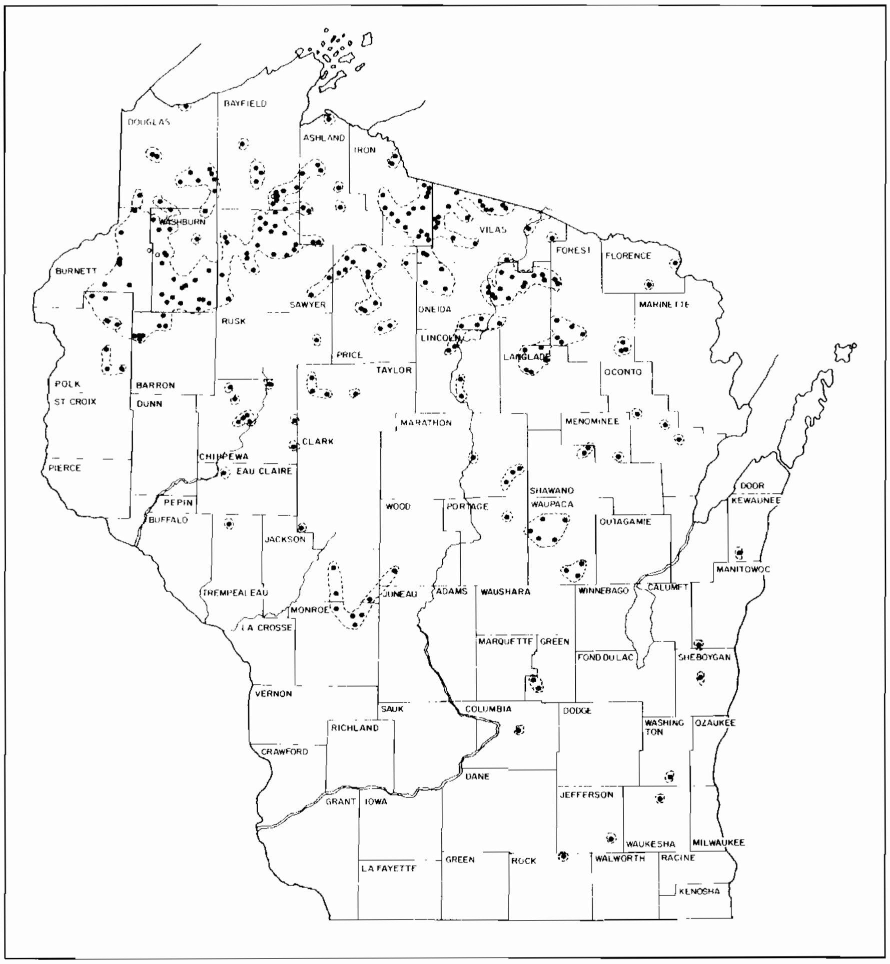


FIGURE 9. *Generalized distribution of "brown lakes" (perceived color), summer 1979 (random data set).*

Hydrogen Ion Concentration (pH)

The hydrogen ion concentration in lakes, expressed in terms of pH units, is largely dependent on the dissociation of acids or bases or their salts to their respective ions. The carbonate-bicarbonate-carbon dioxide system functions as a buffering system which tends to restrict great fluctuations in the pH of lakes. As such, this effect is noted in the relationship of pH and alkalinity, where lakes with very low alkalinities generally have low pH values. pH levels have been shown to have important consequences in aquatic ecosystems. Numerous investigators have reported great differences in species composition and diversity of aquatic plant and animal communities within different pH ranges.

Based on the random data set, 67% of the state's lakes have pH's ranging from 6.5 to 8.0 pH units (Fig. 5). Only 9% of the lakes in the study had surface pH values below 6.0 units at the time of sampling. The mean pH of Wisconsin lakes was 7.15 (median = 7.2), or slightly above neutral (Table 3). The lowest surface pH value (4.3 units) recorded was from Denton Lake, Oneida County. pH is a dynamic parameter in many Wisconsin lakes (Juday, Birge, and Meloche 1935, Lillie and Mason 1980) and is presently the topic of much consideration due to the recent concern over the possible impact of acid deposition on both aquatic and terrestrial ecosystems.

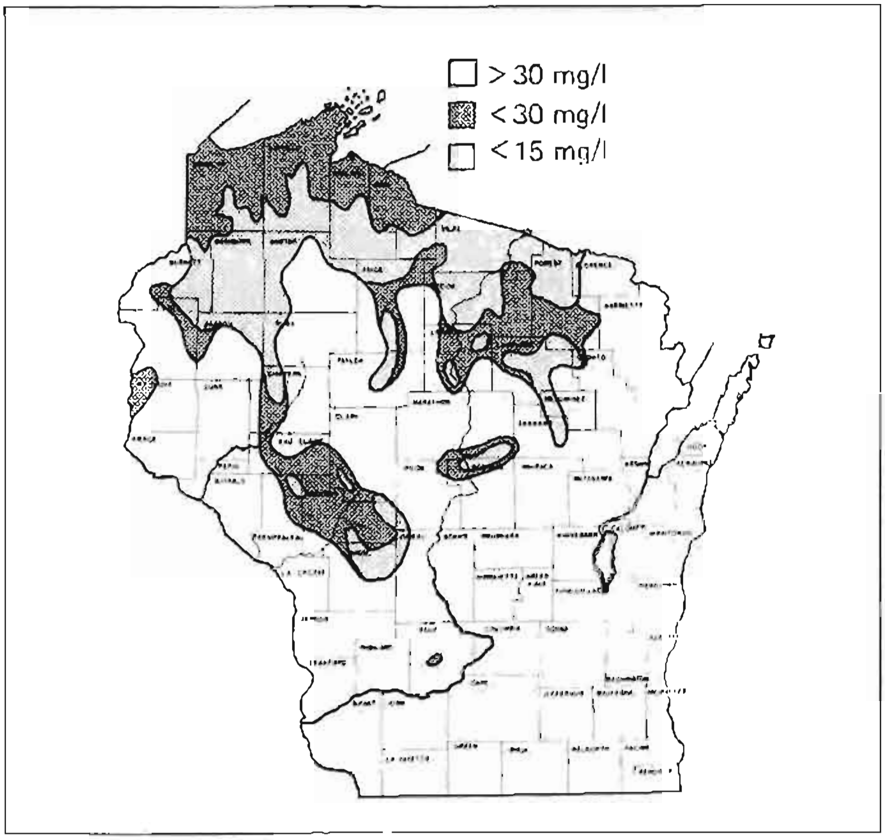


FIGURE 10. *Areas where low (< 15 mg/l) and medium (< 30 mg/l) alkalinity lakes are most likely to be found.*

Calcium and Magnesium

Calcium and magnesium concentrations in Wisconsin lakes are known to be closely related to the bedrock geology of the area (Birge and Juday 1911, Poff 1961). Highest concentrations are associated with the limestone and dolomite deposits of southeastern Wisconsin (Fig. 11). Calcium concentrations (mean = 12 mg/l, median = 8 mg/l) generally exceeded magnesium concentrations (mean = 8 mg/l, median = 2 mg/l)(Table 3), except for slightly higher magnesium in many southeastern Wisconsin lakes. Fifty-five percent of the state's lakes had calcium levels less than 10 mg/l, while 77% had concentrations less than 20 mg/l (Fig. 5). Magnesium levels were below 10 mg/l in 74% of the state's lakes.

Chlorides

Chlorides, of natural origin, may be found in small quantities in almost every lake, and even occur in rainwater. The mean chloride concentration based on the random survey data (4 mg/l, median = 2 mg/l)(Table 3) is nearly identical to the 4.1 mg/l reported for a different set of Wisconsin lakes (Poff 1970) which was not randomly selected. However, the fact that the earlier data set (Poff's 1970 data) was comprised of lakes with relatively higher mean calcium (20.0 mg/l) and magnesium (15.6 mg/l) concentrations than those found in our random data set (calcium = 12 mg/l, magnesium = 8 mg/l) suggests that the chloride levels associated with the relatively lower calcium and magnesium of the random data set are higher than what might be expected. Further evidence supporting this supposition is provided in the Historical section (see increase in chlorides).

Ninety-one percent of the lakes in the random sample had chloride concentrations less than 10 mg/l (Fig. 5). Upper Kelly Lake, Waukesha County, had the highest chloride concentration reported (269 mg/l)(Table 4). Chlorides demonstrated a general increase from north to southeast (Fig. 11). Exceptions (concentration depressions) were found in the central sands areas of Waushara, Marquette and Adams counties and also in smaller pockets of Jefferson, Walworth and Rock counties. Aside from the natural weathering of chloride from bedrock and soils, the major sources of additional chlorides are believed to result from man's activities, specifically the heavy applications of salt in winter road de-icing operations and in effluents from wastewater treatment plants and septic systems.

Sulfate

Sulfate, a naturally occurring ion often associated with heavy mineral deposits, is quite stable once dissolved in water (Hem 1959) and tends to accumulate in the lake ecosystem unless removed. Rainfall may be of ever-increasing importance as a source of sulfate due to the combustion of coal and the subsequent addition of sulfur compounds to the atmosphere. Moyle (1956) discusses the relationship of sulfate concentrations and the distribution of aquatic plants in Minnesota.

Sulfate concentrations were not measured during the 1979 random survey; therefore, the quarterly data set was drawn upon as a source for sulfate information.

Sulfate parallels calcium and magnesium in its statewide distribution (Fig. 11). Highest levels occur in the southeast region and coincide with high population densities and industrialized areas. The small pocket of lakes (mostly Wisconsin River impoundments) in northern Adams and southern Wood counties with higher sulfate concentrations could possibly be associated with industrial activities.

Iron

Wetzel (1975) reported total iron values ranging from 50 to 200 µg/l for the surface waters of hardwater lakes, with higher values expected in lakes with large concentrations of organic matter. Iron concentrations in Wisconsin lakes ranged from less than de-

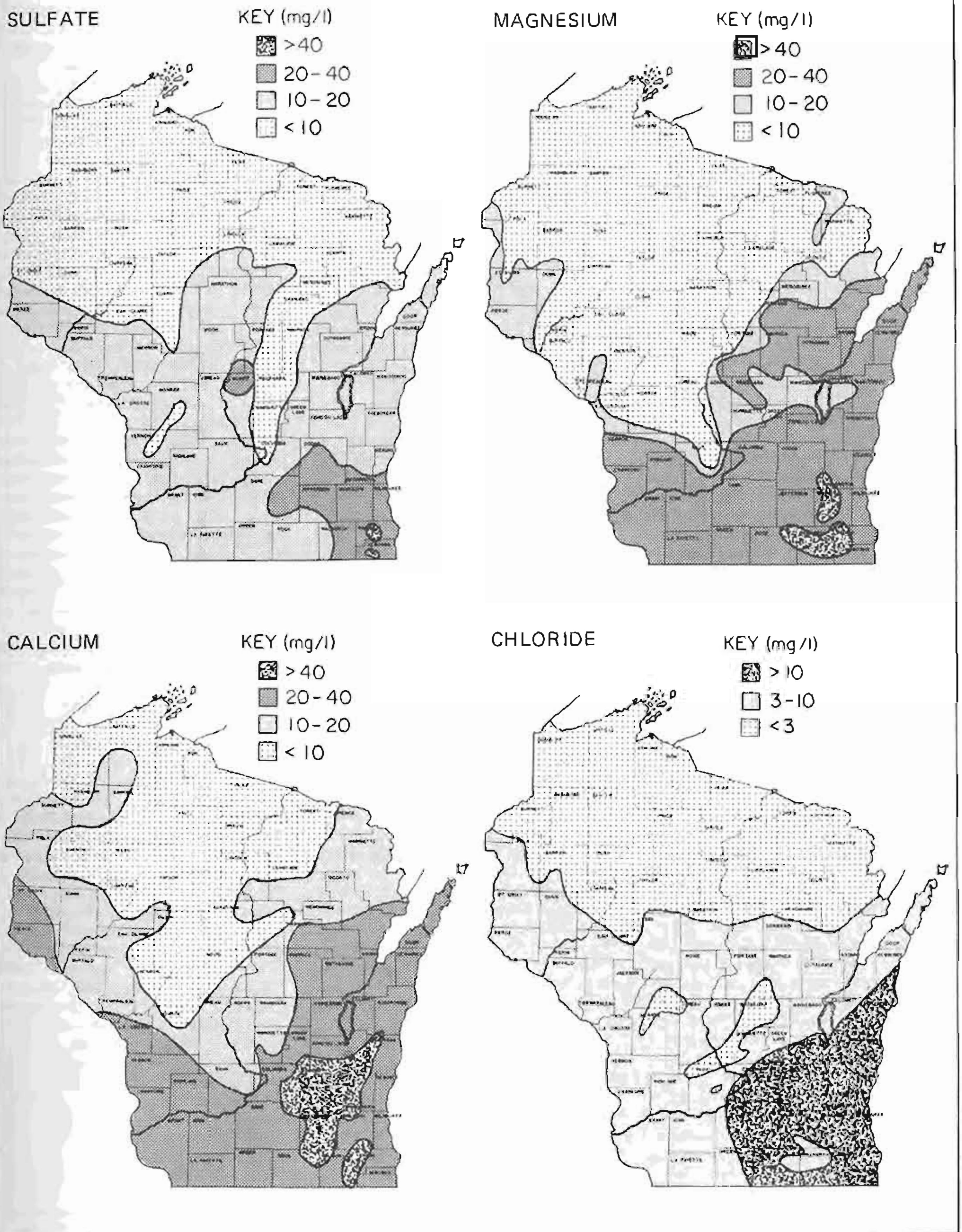


FIGURE 11. *Generalized distributional gradients of selected chemical parameters in the surface waters of Wisconsin lakes (random data set). SO_4 based on quarterly data set.*

tectable (80 μg/l) to as high as 7,200 μg/l in White Sand Lake (T42 R7 Sec 26), Vilas County. There was no meaningful relationship between the presence of high levels of iron in the epilimnetic (oxygenated) waters of Wisconsin lakes and other water quality parameters — even in respect to high color or high turbidity, which was expected based on other studies. Most lakes with high iron concentrations in their surface waters also had high hypolimnetic iron content, but the reverse was not necessarily true. Hypolimnetic iron ranged from 10 to 12,750 μg/l (Pit Lake, Ozaukee County).

Ratios of summer hypolimnetic iron (Fe) and phosphorus (P) have been used in trophic classification (Michalski and Conroy 1972) and are believed to be important indicators of the phosphorus absorbing capacity of lakes. Lakes with high hypolimnetic Fe:P are generally relatively oligotrophic and thought to be capable of absorbing increased phosphorus loading without undergoing a change in trophic status better than lakes with low Fe:P ratios, due to the greater amount of Fe available for precipitating added P.

The latter group of lakes would be expected to have worse epilimnetic water quality than the first-mentioned. A detailed analysis of the Fe:P ratios of the lakes in this study revealed little in support of the theoretical association between Fe:P ratios and lake trophic status. No overall pattern was discerned and values varied considerably within and between counties.

Nutrients — Phosphorus and Nitrogen

Nitrogen and phosphorus are two essential elements contributing to the fertility and growth of plants in lakes. Both elements exist naturally in varying degrees. Nitrogen sources include weathering of bedrock, nitrogen fixation by leguminous plants, the atmosphere, and the natural decay of all forms of plant life. Important additional sources which are attributable to man's activities include livestock wastes, sewage effluents, and applications of agricultural fertilizers.

Phosphorus also occurs naturally in soils and bedrock, but human activities in some watersheds have greatly increased the quantities of phosphorus available to the lake system. Phosphorus sources are generally the same as those listed for nitrogen, with the addition of soap and detergents. A temporary ban on phosphate detergents in effect in Wisconsin since 1979 expired in 1982 when the legislature failed to renew it.

Total phosphorus is a highly variable lake water parameter (see discussion on phosphorus, p. 69) with a wide range in reported values (Tables 3 and 4, Fig. 5). Thermal stratification, settling of organic matter from the epilimnion to the hypolimnion, and release of phosphorus from anoxic sediments within the hypolimnion may account for large differences in the vertical distribution of phosphorus. Variations in epilimnetic phosphorus accompanying seasonal changes will be discussed in a later section. A high percentage (71%) of the lakes in the random data set had total phosphorus concentrations less than 0.03 mg/l (82% less than 0.04 mg/l). The statewide mean total phosphorus value of 0.031 mg/l (median = 0.019 mg/l) was slightly higher than the 0.023 mg/l mean reported for 479 northern Wisconsin lakes by Juday and Birge (1931), which would be expected since the generally more eutrophic southern lakes are included in our data. Highest phosphorus levels were recorded in shallow, turbid lakes or impoundments. Mean total phosphorus for different regions of Minnesota ranged from 0.024 to 0.176 mg/l (Moyle 1954), while Hooper (1956) reported a mean of 0.014 mg/l total phosphorus for eight Michigan lakes. In other areas, Deevey (1940) reported a mean total phosphorus concentration of 0.015 mg/l for 49 Connecticut lakes (0.031 mg/l for lakes of their central lowland region), and Davis et al. (1978) found the mean of 21 Maine lakes to be only 0.008 mg/l.

Organic phosphorus content of Wisconsin lakes averaged 0.017 mg/l, with 89% of the lakes having less than 0.03 mg/l. Inorganic phosphorus averaged 0.013 mg/l, with 88% of the lakes having less than 0.02 mg/l. Organic phosphorus was the predominant form (greater than 50%) in over 72% of the lakes in the sample.

The mean percentage of inorganic phosphorus for all randomly sampled lakes (39%) was very similar to the 41% orthophosphorus reported by Omernik (1977) for streams draining watersheds of similar nature to Wisconsin's general land use.

Total nitrogen is a relatively stable constituent as compared to phosphorus, with much lower coefficients of variation (see discussion p. 69). Moyle (1954) reported total nitrogen means for different Minnesota lake regions ranging from 0.234 to 1.22 mg/l, with absolute values ranging from 0.06 to 5.92 mg/l. Wisconsin values are very similar — the overall mean was 0.86 mg/l (median = 0.73 mg/l), and the range was 0.05 to 8.46 mg/l (Tables 3 and 4). Seventy-one percent of the lakes fell within the 0.30 to 1.00 mg/l total nitrogen range (Fig. 5).

Inorganic nitrogen averaged 0.26 mg/l with 70% of the lakes having less than 0.30 mg/l. Organic nitrogen averaged 0.60 mg/l and 78% of the lakes had between 0.2 and 0.8 mg/l. Organic nitrogen exceeded the inorganic fraction in over 94% of the lakes sampled, which would be expected during the summer. On the average, 74% of the nitrogen present was in the organic form.

Some interesting comparisons can be made with Omernik's (1977) data on mean nutrient concentrations for streams draining various types of watersheds throughout the United States. Omernik's reported means for streams draining 295 watersheds with over 50% forest cover were: for orthophosphorus, 0.014 mg/l; total phosphorus, 0.034 mg/l; inorganic nitrogen, 0.294 mg/l; and total nitrogen, 0.839 mg/l. These values are very similar in both absolute value and percentage to the means (0.013 mg/l, 0.031 mg/l, 0.256 mg/l, and 0.86 mg/l, respectively) reported for Wisconsin lakes, where forest cover makes up approximately 45% of the state's land surface cover (Stone and Thorne 1961). Considering the fact that most of Wisconsin's lakes are located in the more heavily forested "northwoods" country, it is not too surprising that the mean nutrient values closely correspond to the reported means for streams draining watersheds with greater than 50% forested lands.

Dissolved Oxygen

Since dissolved oxygen plays a significant role in chemical and biological processes in lakes, the amount found in lake waters is of great importance. Dissolved oxygen is an especially critical factor affecting aquatic biota such as fish and aquatic insects; Wisconsin's water quality standards specify 3 and 5 mg/l dissolved oxygen as required to support warm- and cold-water game fish species, respectively. Also, a biotic index has been developed in Wisconsin for rating water quality of streams using empirical oxygen tolerance levels (relative abundance) of various species of aquatic invertebrates as a major consideration (Hilsenhoff 1982).

In our study, several thousand dissolved oxygen profiles were run on Wisconsin lakes during all seasons of the year. Lakes were generally saturated with oxygen throughout the water column in spring and fall, with the exception of a few lakes which did not experience turnover. Likewise, oxygen concentrations in the epilimnetic waters of lakes during summer were always found to be adequate (> 3 mg/l) for warm-water fish life. However, epilimnetic dissolved oxygen was measured in the pelagic zone during daylight hours; dissolved oxygen depletion and summerkill of fish have been observed on rare occasions in highly productive lakes or stagnant bays where photosynthetic and respiration rates are high.

Many Wisconsin lakes that thermally stratify in summer and winter lose some or all of their hypolimnetic dissolved oxygen before the end of the stagnation period. The amount of oxygen found in the hypolimnion at the time of sampling depends primarily on the oxygen depletion rate, the oxygen content of the lake at the end of the turnover period, and the volume of the hypolimnion in relation to the influx of nutrients (Hutchinson 1938, Stewart 1976). In the random survey, sampling did not begin until summer stratification had been in progress for a considerable length of time (early July). Approximately 50% of the lakes deep enough to thermally stratify, or 24% of all the lakes sampled (random survey), exhibited severe reduction of dissolved oxygen concentrations in the hypolimnion (Table 5).

Hypolimnetic dissolved oxygen depletion has often been associated with the eutrophication process in lakes (Hutchinson 1938, Deevey 1940, Hasler 1947). Our study further demonstrated the association between oxygen content of the hypolimnion and trophic status; 87% of all stratified lakes with epilimnetic chlorophyll *a* levels over 15 µg/l had low dissolved oxygen concentrations in the hypolimnion.

In winter, reduction of dissolved oxygen concentrations under ice cover to such low levels that fish die-offs occur is popularly referred to as "winterkill". The mechanisms involved in the development of conditions leading to winterkill in lakes have been extensively investigated (Greenbank 1945, Welch, Dillon and Screedharan 1976, Mathias and Barica 1980), and an excellent report on the problem of winterkill is presented by Schneberger (1970). A wide range of factors contribute to the conditions causing winterkill, but certain factors appear more important than others: mean or maximum depth, lake trophic status (extent of weed and algae growth), sediment area:lake volume ratio (Mathias and Barica 1980), and climatic conditions influencing ice and snow cover (Greenbank 1945).

Our data reflect only the conditions existing in the surveyed lakes at the time of winter quarterly sample collection, and may not represent "worst case" conditions. In many lakes, winterkill conditions do not develop each year, or may develop only for short periods of time in any given year; therefore, low dissolved oxygen conditions could have been missed by sampling crews. Lakes which had dissolved oxygen levels less than 1 mg/l in the entire water column were considered "severe" cases where winterkill was a possibility (in most cases it was not determined whether winterkill actually occurred). Eight percent of the 535 lakes where winter data were collected had severe dissolved oxygen conditions, and another 18% had moderate stress conditions (< 1 mg/l somewhere in the water column). The characteristics of winterkill lakes are discussed under Dissolved Oxygen Conditions, pp. 31-32.

LAKE TYPES

Characterization of individual lakes generally is not dealt with in this report, although data are shown for selected lakes to illustrate historical trends. Rather, this section will present general characteristics of *groups* of lakes which are considered to be of a similar nature. These lakes are grouped on the basis of one or two common characteristics (e.g., all seepage lakes that are thermally stratified vs all drainage lakes that are completely mixed) for the purpose of (1) identifying "normal" or average water quality existing in each lake type, (2) making comparisons between different types of lakes, and (3) determining the geographical distribution of particular lake types with certain water quality characteristics.

The question of which factor or factors are most important in determining the overall water quality of a lake is extremely difficult to assess. While lakes may be grouped on the basis of a similar physical or chemical characteristic, they may be totally different in many other respects. For example, all drainage lakes have an outlet flow as a common characteristic, but they vary widely in size, depth, shape and numerous other characteristics. These differences among drainage lakes have profound and sometimes offsetting effects on in-lake water quality. Thus, deep drainage lakes may more closely resemble deep seepage lakes than shallow drainage lakes.

Comparisons between lakes classified according to one particular characteristic may be greatly influenced by the numerical composition of individual lake types based on some other perhaps more important water quality characteristic within the sets. Therefore, in the case of the drainage lake subset used as an example above, the greater the number of thermally stratified lakes there are in the subset, the better the overall water quality will be and the less difference there will be between the drainage and seepage subsets. Conversely, the greater the number of mixed lakes there are in the drainage lake subset, the poorer the water quality will be and the greater the differences between the drainage and seepage subsets. This impact is unavoidable and "randomized" by the use of the random data set. Cluster analysis of the data was not carried out but could possibly give some further insight into the relative importance of various factors influencing lake water quality.

Natural Lakes vs Impoundments

Most water quality characteristics of natural lakes are quite distinct from those of impoundments (Table 6, Append. A). Lakes generally have lower nutrient concentrations than impoundments and therefore generally have better overall water quality. Water clarity measurements were distinctly better in natural lakes than in impoundments, but chlorophyll *a* levels (means) were not significantly different at the 95% C.I. level (Append. A). However, at the 90% C.I. level, chlorophyll *a* concentrations were generally lower in natural lakes. Lakes had lower total nitrogen levels than impoundments but similar organic nitrogen levels, which may indicate that natural lakes more efficiently convert inorganic nitrogen to organic nitrogen. This apparently results from a wide combination of factors including different morphometry, smaller watersheds, generally longer retention times, and better water clarity.

Clear distinctions were evident in total phosphorus concentrations between lakes and impoundments (Append. A). Differences in the inorganic-organic phosphorus fraction between lakes and impoundments were also found; organic phosphorus made up 60% of the total phosphorus in lakes and only 46% of the total phosphorus in impoundments.

While impoundments can be characterized as generally having higher nutrient content and poorer water quality than natural lakes, many Wisconsin impoundments, some large ones in particular, are recognized as outstanding fish producers.

Drainage Type

While all impoundments by definition have outlet flows, and are thus categorized as drainage lakes, natural lakes may be of either drainage (outlet) or seepage (no outlet) type. Our drainage classification system refers only to the existence of an outlet flow and does not consider the source of water entering the lake. However, most drainage lakes also have inlet flows, while most

TABLE 6. *Statistical summary of major characteristics of natural lakes and impoundments (random data set).*

Parameter	Value	No. Lakes	Mean	Standard Deviation	Median
NATURAL LAKES					
Area	acres	558	176	530	69
Mean depth	ft	185	13.5	7.5	12.6
Maximum depth	ft	557	27	19	22
Color	units	480	35	36	20
Transparency	m	505	2.4	1.4	2.1
Chlorophyll *a*	µg/l	541	13.4	40.8	7.1
Chlorides	mg/l	507	4	6	1
Calcium	mg/l	505	10	10	7
Magnesium	mg/l	505	7	9	2
pH	units	557	7.1	0.86	7.1
Alkalinity	mg/l	557	45	50	26
Turbidity	JTUs	542	2.9	4.6	2.0
Organic N	mg/l	556	0.59	0.36	0.52
Total N	mg/l	556	0.82	0.57	0.69
Inorganic P	mg/l	555	0.010	0.021	0.004
Total P	mg/l	556	0.025	0.031	0.018
IMPOUNDMENTS					
Area	acres	100	630	2,446	132
Mean depth	ft	43	5.6	2.6	5.1
Maximum depth	ft	99	15	9	12
Color	units	77	65	52	55
Transparency	m	88	1.3	0.7	1.2
Chlorophyll *a*	µg/l	99	22.3	27.2	11.0
Chlorides	mg/l	96	7	9	4
Calcium	mg/l	96	22	18	16
Magnesium	mg/l	96	14	14	8
pH	units	100	7.5	0.7	7.5
Alkalinity	mg/l	100	92	79	64
Turbidity	JTUs	100	4.2	3.9	3.0
Organic N	mg/l	100	0.65	0.35	0.60
Total N	mg/l	100	1.06	0.54	0.94
Inorganic P	mg/l	100	0.035	0.077	0.010
Total P	mg/l	100	0.064	0.100	0.035

TABLE 7. *Statistical summary of major characteristics of seepage and drainage lakes, including impoundments (random data set).*

Parameter	Value	No. Lakes	Mean	Standard Deviation	Median
SEEPAGE LAKES					
Area	acres	332	93	142	51
Mean depth	ft	95	13.4	8.0	11.5
Maximum depth	ft	332	26	18	21
Color	units	310	27	29	15
Transparency	m	288	2.7	1.5	2.4
Chlorophyll *a*	µg/l	319	11.3	34.3	6.3
Chlorides	mg/l	303	3	5	1
Calcium	mg/l	302	8	9	4
Magnesium	mg/l	302	5	9	1
pH	units	331	6.8	0.9	6.8
Alkalinity	mg/l	331	34	48	11
Turbidity	JTUs	328	2.3	1.8	1.9
Organic N	mg/l	330	0.55	0.33	0.46
Total N	mg/l	330	0.76	0.57	0.64
Inorganic P	mg/l	329	0.008	0.013	0.004
Total P	mg/l	330	0.021	0.028	0.015
DRAINAGE LAKES					
Area	acres	328	398	1,509	118
Mean depth	ft	134	11.1	7.0	9.2
Maximum depth	ft	327	25	19	18
Color	units	250	54	47	42
Transparency	m	307	1.9	1.2	1.5
Chlorophyll *a*	µg/l	324	18.2	43.1	8.7
Chlorides	mg/l	303	5	8	3
Calcium	mg/l	302	17	14	12
Magnesium	mg/l	302	10	12	4
pH	units	329	7.5	0.7	7.5
Alkalinity	mg/l	329	71	63	48
Turbidity	JTUs	317	4.0	6.1	2.4
Organic N	mg/l	329	0.65	0.37	0.58
Total N	mg/l	329	0.95	0.55	0.83
Inorganic P	mg/l	329	0.019	0.049	0.005
Total P	mg/l	329	0.040	0.064	0.025

seepage lakes are fed primarily either by groundwater or diffuse overland surface flows.

Based on comparisons of means, seepage lakes have generally better water clarity and are less eutrophic than drainage lakes (Table 7, Append. A). While confidence intervals of chlorophyll *a* means overlap, seepage lakes have somewhat lower levels (86% less than 15 µg/l) than drainage lakes (73% less than 15 µg/l). Seepage lakes also generally have lower color than drainage lakes (Fig. 11), which agrees with the earlier studies on the color of Wisconsin lake waters by Juday and Birge (1933) (also see Schindler 1971*a*). Sixty-four percent of the randomly sampled lakes with measured color values above 40 units were of the drainage type (Fig. 13). Most seepage lakes with high color were bog lakes.

Mean and maximum depths of seepage and drainage lakes in this study were not significantly different. However, considerable difference in water chemistry was evident (Append. A). Seepage lakes had significantly lower levels of nutrients and lower mean pH and alkalinity than drainage lakes; mean alkalinity (34 mg/l) of seepage lakes was the lowest for any subgroup of lakes. Seepage lakes also had much smaller drainage basin:lake area ratios, which might help account for the lower nutrient levels.

Lake Morphometry and Thermal Stratification

Lake morphometry has long been recognized as having important influence on the dynamic processes within individual lakes (Birge and Juday 1911, Deevey 1940, Hutchinson 1957, Hayes 1963, Hayes and Anthony 1964, Schindler 1971*a* and 1978, Kerekes 1975). Mean depth is considered to be among the more important morphological factors influencing the water quality conditions in a lake (Rawson 1952 and 1955, Sakamoto 1966, Vollenweider 1968, Ryder et al. 1974, Schneider 1975). There is, however, a general lack of agreement as to which factors, if any, are of greatest significance in determining overall water quality. There are several important considerations which prohibit such an assessment. Fee (1979) has demonstrated the importance of the epilimnetic volume in relation to in-lake nutrient recycling and resulting water quality. Differences in retention times, percentage of bottom area exposed to epilimnetic recycling vs the total lake area, the volume of the hypolimnion functioning as a temporary nutrient trap, and perhaps most importantly, the differences in external nutrient loading to the lake (Schindler 1978) are all factors which limit the possibilities of comparing depth with in-lake water quality.

Nevertheless, it may be generally stated that lakes with greater mean and maximum depths have significantly better water quality than lakes with lower mean and maximum depths (Append. A). Further discussion concerning the relationship of depth to other water quality parameters may be found in the Interrelationships section, p. 79.

Maximum depth, which is highly correlated with mean depth, becomes increasingly important to overall water quality when a lake is deep enough to thermally stratify. The maximum and mean depths, together with the shape and area of the lake basin, determine the volume and/or area of epilimnetic water exposed to bottom sediment contact and subsequent resuspension of nutrient-rich seston. These factors also control the volume of water in the hypolimnion, which acts as a temporary nutrient trap for seston and particulates "raining" down from the epilimnion. The epilimnetic water quality is highly dependent upon these physical features of the lake basin.

Stratification is dependent upon several physical features in addition to maximum depth including: area, retention time, basin shape, water color, orientation to the prevailing winds, and surrounding topographical features. Hutchinson (1957) reviews the processes involved in stratification originally presented by Birge (1916). A model for predicting thermal stratification of Wisconsin natural lakes given surface area and maximum lake depth, developed from data obtained during the quarterly sampling program, is presented elsewhere (Lathrop and Lillie, 1980).

Whether or not a lake thermally stratifies is of paramount importance in determining how it responds to the influx of nutrients. While it appears that loadings to both stratified and mixed lakes were not significantly different (implied by the fact that mean alkalinities, calcium, magnesium, chloride and pH were quite similar between mixed and stratified lakes — see Append. A), water transparencies, chlorophyll *a* and nutrient concentrations in the epilimnetic waters of stratified lakes were generally significantly lower than in mixed lakes (Table 8, Append. A). These similar differences in water transparencies, chlorophyll *a* and nutrients noted in the case of Lakes vs Impoundments and Seepage vs Drainage may simply be attributable to differences in loadings as indicated by differences in alkalinities, calcium, magnesium, etc.

TABLE 8. *Statistical summary of major characteristics of mixed and thermally stratified lakes, including impoundments (random data set).*

Parameter	Value	No. Lakes	Mean	Standard Deviation	Median
MIXED LAKES					
Area	acres	261	304	1,599	70
Mean depth	ft	113	6.6	3.4	6.0
Maximum depth	ft	261	12	6	11
Color	units	229	41	40	25
Transparency	m	206	1.6	0.9	1.4
Chlorophyll *a*	µg/l	249	21.7	60.2	9.2
Chlorides	mg/l	260	4	7	2
Calcium	mg/l	259	13	15	8
Magnesium	mg/l	259	8	11	3
pH	units	260	7.2	0.8	7.2
Alkalinity	mg/l	260	60	68	34
Turbidity	JTUs	258	4.2	6.6	2.6
Organic N	mg/l	260	0.66	0.40	0.56
Total N	mg/l	260	1.01	0.72	0.84
Inorganic P	mg/l	259	0.022	0.055	0.008
Total P	mg/l	260	0.043	0.074	0.021
THERMALLY STRATIFIED LAKES					
Area	acres	282	214	669	76
Mean depth	ft	156	16.9	8.0	15.4
Maximum depth	ft	282	39	18	35
Color	units	228	33	30	20
Transparency	m	280	2.7	1.4	2.7
Chlorophyll *a*	µg/l	278	10.4	13.3	6.1
Chlorides	mg/l	282	4	7	2
Calcium	mg/l	281	12	11	9
Magnesium	mg/l	281	8	11	3
pH	units	283	7.2	0.8	7.3
Alkalinity	mg/l	283	56	55	37
Turbidity	JTUs	280	2.4	2.2	1.9
Organic N	mg/l	282	0.53	0.30	0.46
Total N	mg/l	282	0.74	0.41	0.65
Inorganic P	mg/l	282	0.009	0.012	0.004
Total P	mg/l	282	0.023	0.023	0.017

Lake Size: Wisconsin's Largest Lakes and Impoundments

In Wisconsin, the 127 lakes that are 1,000 acres in size or larger comprise 54% of Wisconsin's total lake surface acreage and represent a very important asset in terms of their recreational and economic values. The water quality of this group of lakes is of considerable importance to a great many Wisconsin citizens, since these lakes receive very heavy public use. A great diversity of lake types and water quality characteristics are found within the 1,000-acre lake group, but because 36% are impoundments and 55% are drainage lakes, they have generally poorer-than-average water quality as a group (Append. A). Average chlorophyll *a* levels (22.1 µg/l), total phosphorus concentrations (0.070 mg/l), and water clarity measurements (2.0 m) indicate somewhat poorer-than-average conditions, compared to all lakes sampled in the random survey (Tables 3 and 9). Although trophic classification levels of the larger Wisconsin lakes range from oligotrophic to highly eutrophic, relatively high levels of nutrients, turbidities, alkalinities, calcium, magnesium and chlorides are usually present (Table 9).

Drainage Basin Size: Lake Area Ratio and Retention Time

The ratio of a lake's drainage basin area to its surface acreage has a significant impact on water quality, reflected by differences in color, chloride, calcium, magnesium, pH, alkalinity, turbidity, total nitrogen, inorganic and total phosphorus, and water clarity (Append. A). Generally, the larger the drainage basin:lake area (DB:LA) ratio the higher the concentration of these particular water quality parameters. This is believed to be primarily the result of increased input of materials to the lake basin on an areal basis. Impoundments and mixed drainage lakes generally have higher DB:LA ratios than seepage lakes or stratified drainage lakes (Tables 10 and 11), which is one of the reasons why they

TABLE 9. *Characteristics of 1,000⁺-acre lakes (total data set).**

Parameter	Value	No. Lakes	Mean	Standard Deviation	Median
Area	acres	127	4,087	12,398	1811
Mean depth	ft	93	17.5	13.4	14
Maximum depth	ft	127	42	33	36
Color	units	68	38	24	40
Transparency	m	122	2.0	1.4	1.5
Chlorophyll *a*	μg/l	81	22.1	28.9	20.7
		(24)	(17.1)	(20.5)	(7.6)
Chlorides	mg/l	126	7	10	3
Calcium	mg/l	126	18	12	13
Magnesium	mg/l	126	11	12	6
pH	mg/l	127	7.6	0.6	7.5
Alkalinity	mg/l	127	75	59	47
Turbidity	JTUs	117	4.3	4.5	2.3
Organic N	mg/l	127	0.71	0.49	0.62
Total N	mg/l	127	1.03	0.69	0.87
Inorganic P	mg/l	127	0.038	0.084	0.017
Total P	mg/l	127	0.070	0.125	0.036
Drainage basin: lake area	ratio	121	236	7	14
Retention time in years	years	88	1.9	2.6	0.9

*Random data statistics in ( ).

TABLE 10. *Mean retention times and drainage basin size: lake area ratios (DB:LA) for impoundments and natural lakes (total data set).*

Lake Type	DB:LA	Retention Time (years)
Impoundments	676	0.12
Seepage lakes	8	2.15
Drainage lakes	88	1.42

TABLE 11. *Comparison of mean drainage basin size:lake area ratios (DB:LA) and retention times for mixed and stratified seepage and drainage lakes (total data set).*

	DB:LA		Retention Time (years)	
Lake Type	Mixed	Strat.	Mixed	Strat.
Seepage lakes	7	9	1.24	2.63
Drainage lakes*	196	39	0.51	1.92

*Includes impoundments.

generally have poorer water quality.

Lakes with large DB:LA ratios normally have short hydraulic retention times and vice versa (Tables 10-12). Seepage lakes generally have smaller DB:LA ratios and longer retention times than drainage lakes. A linear regression equation was developed for 307 lakes in our random data set that can be used to predict a lake's retention time given the DB:LA ratio (Fig. 12). This line is similar to that developed previously by Bartsch and Gakstatter (1978) for a number of northern U.S. lakes. Differences between the lines probably can be attributed to different runoff coefficients (which are dependent on climatic conditions, vegetation, soils and topography of the watershed) and to differences in lake basin morphometry (greater volumes per unit surface area) for Wisconsin lakes.

The impact of retention time on water quality as expressed by total phosphorus concentrations can be observed through the close associations of retention time with DB:LA and mean depth (Table 12). Mean and median total phosphorus concentrations decrease with increasing retention times, but it is not clear whether this effect is due to (1) retention, (2) the increase in mean depth, (3) the decrease in DB:LA, or (4) a combination of all three factors. Bartsch and Gakstatter (1978) emphasized that a long hydraulic flushing time functions to increase the proportion of incoming nutrients that will be removed from the lake system by sedimentation. They stated that in comparing two lakes of similar size and watershed acreage (and assuming equal nutrient loadings), the deeper lake will have the greater volume (greater dilution effect) and theoretically the longer retention time (greater sedimentation rate), and therefore should have the better water quality of the two.

The Bartsch and Gakstatter theory is that lakes with long retention times tend to accumulate a greater percentage of incoming nutrients than lakes of similar size, drainage basin size and nutrient loading but shorter retention times. This means some lakes reach a "saturation point" where, due to their physical morphometry and trophic status, undesirable water quality conditions are created. At that point, the nutrient loading in combination with the lake's morphometric characteristics (which control internal nutrient recycling) are such that a considerable deterioration of surface water quality occurs, as evidenced by poorer water clarity and increased chlorophyll *a* levels.

Garn and Parrott (1977) have carried nutrient loading-retention time considerations further by computing the trophic response rate or lake "sensitivity" to changes in nutrient loading for a number of national forest lakes, based on initial trophic state and hydrologic and morphologic characteristics. In general, lakes with short retention times may be expected to have fast response rates, while lakes with long retention times should have slow response rates. This could be of great significance in lake management, because lakes with long retention times, while being the least sensitive to inputs of nutrients, may also be the slowest to respond to a decrease in nutrient input (Dillon and Rigler 1975).

Too many assumptions, variables and unknowns exist to adequately assess these theories based on our data. Most retention times used in this report were calculated based on areal runoff coefficients, watershed size and lake volume; therefore, there could be substantial error in estimates of retention time for some lakes, especially seepage lakes where groundwater inflow and outflow rates might be inaccurate.

Also, important factors such as biological interactions and climatic influences preclude evaluation of these hypotheses based on our data.

TABLE 12. *Selected characteristics of lakes within different hydraulic retention time ranges (total data set).*

	Retention Time (days)					
	0-14	14-60	61-180	181-365	365-730	>730
Area (acres)						
No.	79	70	72	129	135	153
Mean	558	1,547	504	1,507	306	762
SD*	1,480	3,928	972	12,111	629	1,317
Median	161	292	164	156	129	270
Mean Depth (ft)						
No.	79	70	73	129	135	153
Mean	5.7	8.0	10.6	11.0	13.4	22.8
SD	3.0	4.8	6.5	6.3	5.7	10.9
Median	5.2	7.2	9.0	9.6	12.4	21.1
Maximum Depth (ft)						
No.	80	70	73	129	135	153
Mean	15.6	21.1	25.2	27.1	35.3	57.0
SD	10.4	15.7	16.7	18.0	19.0	28.0
Median	12.0	17.0	21.0	22.0	32.0	52.0
Total Phosphorus (mg/l)**						
No.	79	69	73	123	135	150
Mean	.094	.085	.056	.048	.033	.025
SD	.079	.161	.119	.063	.037	.021
Median	.075	.040	.030	.030	.024	.020
DB:LA Ratio						
No.	80	70	73	129	135	153
Mean	1,166	142	42	15	8	5
SD	1,352	134	32	13	6	3
Median	666	119	29	11	6	4

*Standard deviation.
**Summer values.

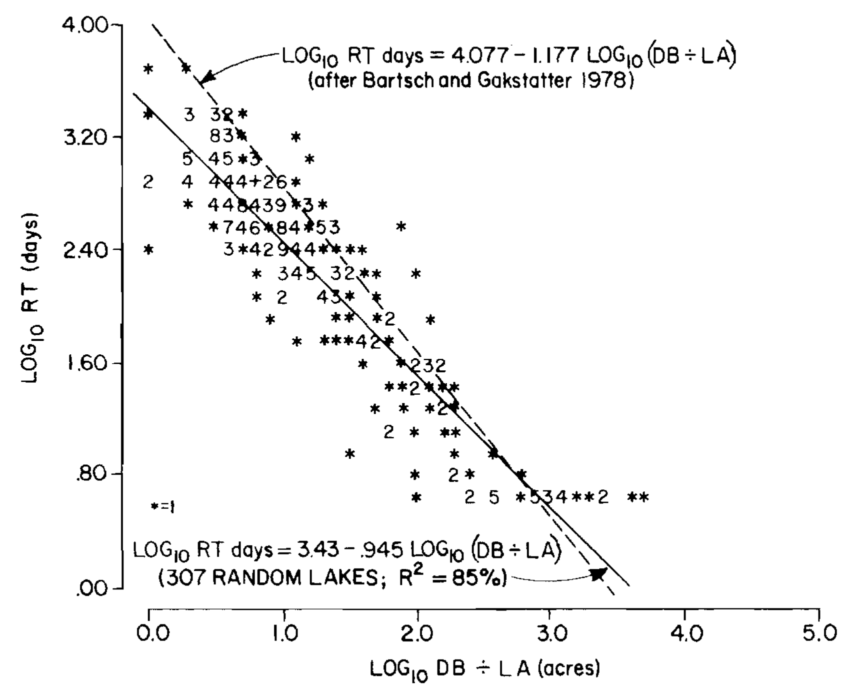


FIGURE 12. *Relationship of retention time and DB:LA ratios (307 lakes) (random data set). Note: Morphometric information for all lakes in the random data set was not available; thus, some bias may be present in the relationship.*

Color

All but 2 of the 124 lakes with measured color less than 10 units were natural lakes. By contrast, 97% of the randomly sampled impoundments had color levels in excess of 10 units (Fig. 8). A majority of Wisconsin lakes with high color were drainage lakes (Figs. 8, 13). In the plot of numerical distribution of lakes based on measured color (Fig. 13), a small second peak is observed in the frequency distribution similar to that reported by Juday and Birge (1933), but its significance is unknown.

Chlorophyll *a* levels in highly colored lakes (> 100 units) were no greater than those in lakes with low color (< 40 units). While these low color lakes had slightly higher turbidities than the high color lakes, water clarity was much reduced in the high color lakes suggesting that color was the principal factor involved in the reduction (Table 13). Eighty-one percent of the highly colored lakes were perceived as "brown" by field observers (Fig. 14).

To many people, one of the most important characteristics of lakes is aesthetic appearance, or "how they look". Good scenery and clean water are highly regarded amenities of lakeshore property owners (Klessig 1973, Smith and Mulamoottil 1979). Smith and Mulamoottil (1979) found a strong relationship between length of ownership and the owners' perception of the importance of water color. Short-term lakeshore property owners indicated their lakes had poor water color, implying either that lake frontage development progressed from clean, clear lakes to more highly colored lakes or that property on lakes with poor water color exchanged hands more frequently.

The fact that perceived and measured water color do not necessarily coincide is indicated in the results of this study, since the mean measured color of lakes classified as "green" by field investigators (24 units) was identical to that for lakes labeled as "blue" (clear)(Fig. 15).

Raman (1922) indicated that the green or bluish-green appearance of water may be due to the scattering of light caused by the presence of suspended matter. This appears to be the case with "green" lakes in our survey, as indicated by the high chlorophyll *a* means (27 μg/l) of the green lakes (Fig. 15).

Green lakes can be characterized as having relatively poor water clarity, moderately high alkalinity, low measured color, and high total phosphorus, pH, turbidity and chlorophyll *a* (Fig. 15). Low alkalinity (< 15 mg/l) green lakes have significantly higher pH values than clear or brown lakes

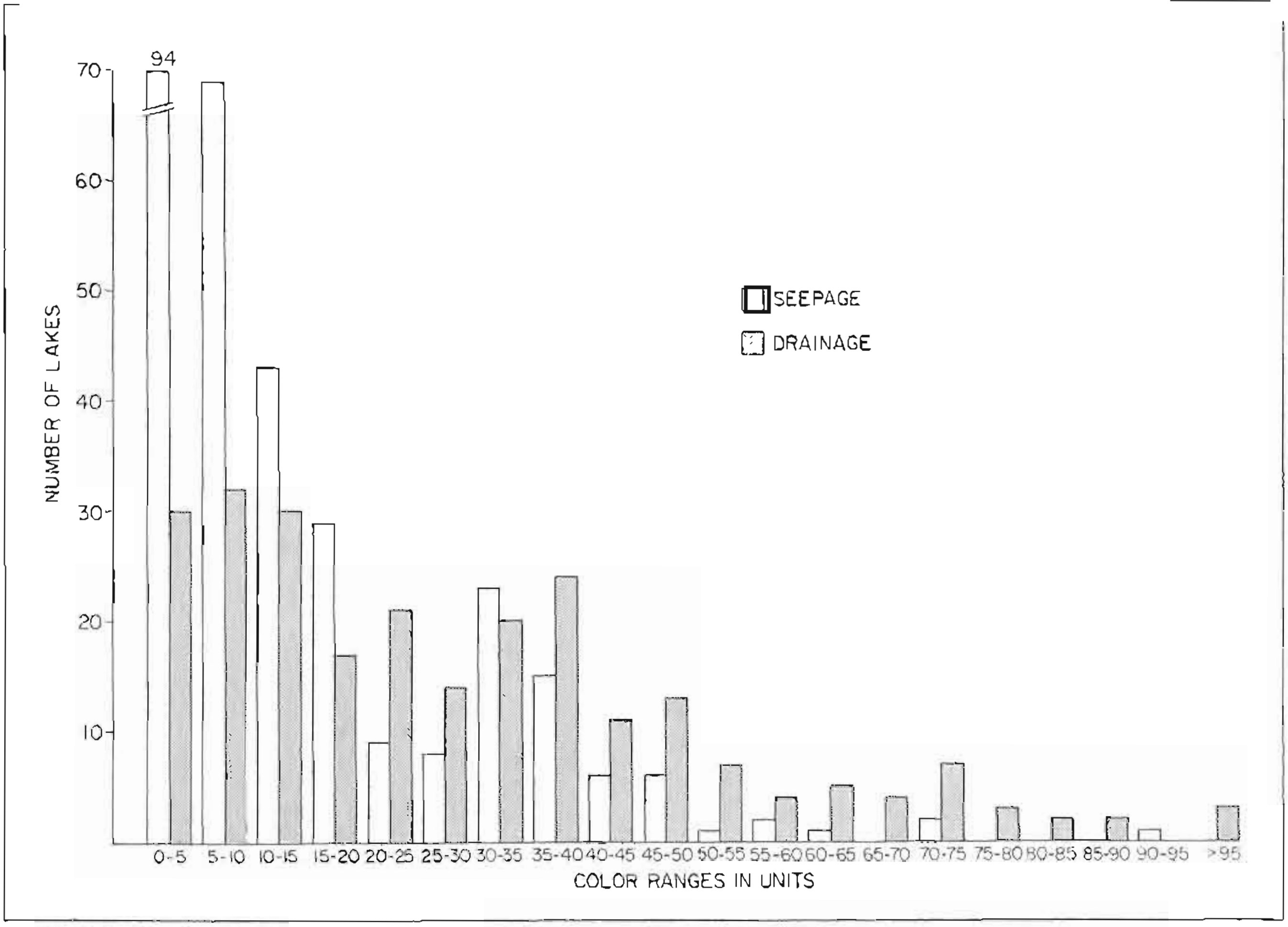


FIGURE 13. *Number of seepage and drainage lakes in various color ranges (random data set).*

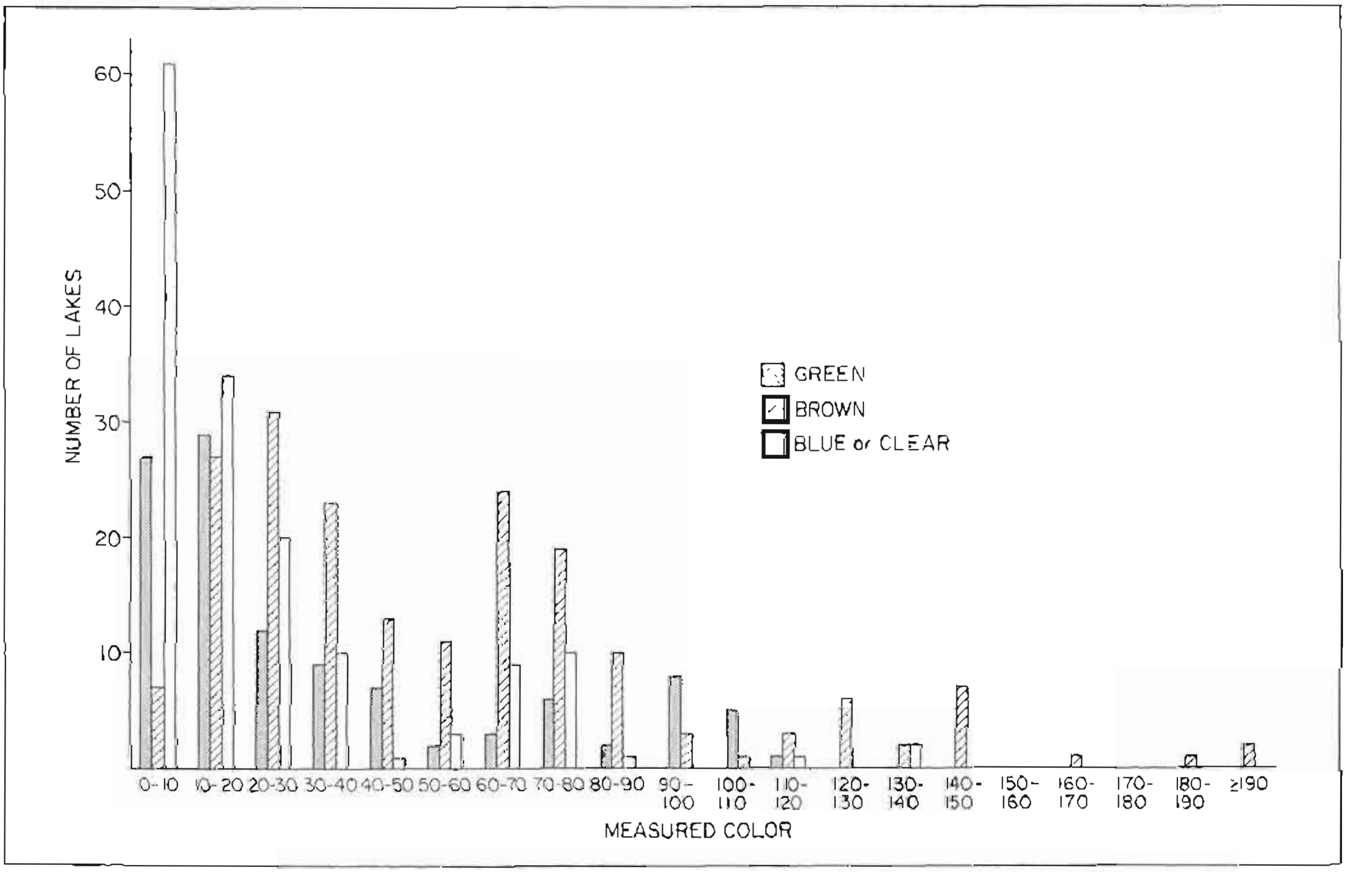


 FIGURE 14. *Measured vs perceived color of lakes (random data set).*

with similar alkalinities (Fig. 16). The majority of high chlorophyll *a* lakes were green in appearance; however, the majority of green lakes had chlorophyll *a* levels less than 15 μg/l (Figs. 17 and 18).

Turbid lakes have below-average water quality with high total phosphorus, pH, measured turbidity, alkalinity, color and poor water clarity (Fig. 15). Chlorophyll *a* levels varied considerably, but averaged 13 μg/l.

Brown lakes are characterized by their high measured color levels, moderate alkalinities, fairly low turbidities, low pH, moderate phosphorus and chlorophyll *a* concentrations, and generally poor water clarity (Fig. 15). The few brown lakes with chlorophyll *a* levels greater than 15 μg/l may have been perceived as brown due to large concentrations of diatoms which often impart a cloudy brown appearance to lake waters.

All 177 lakes described as either clear or blue had chlorophyll *a* concentrations less than 15 μg/l, and 94% had chlorophyll *a* levels less than 10 μg/l (Figs. 17, 18). Most of the clear lakes with chlorophyll *a* levels between 10 and 15 μg/l were quite shallow and therefore the color or composition of the lake bottom may have interfered with the perception of water color. The mean chlorophyll *a* concentration of clear lakes, 5 μg/l (Fig. 15), i ndicates that a majority of these lakes are either oligotrophic or borderline between oligotrophic and mesotrophic classifications (Table 28). The mean total phosphorus value for "clear" lakes (0.015 mg/l) falls in the middle of the range of values used to separate lakes by trophic status on the basis of phosphorus content. Clear lakes had moderate alkalinities and pH values, low turbidities and color levels, and very good water clarity (Fig. 15).

The impact of color on water clarity is best illustrated by comparison of the relationship between color type and Secchi disc transparency. Brezonik (1978) computed theoretical maximum water clarities for lakes with various levels of organic color (Fig. 19). For Wisconsin lakes, the difference in water clarity between the clear lakes and the green and brown lakes is due to chlorophyll *a* and/or inorganic turbidities which reduce water clarity. However, the divergence of our data from Brezonik's and from those reported by Juday and Birge (1933) at the lower color levels cannot be explained. The divergence appears to begin at color levels at or below 75 units and is similar for all three perceived color groups. Atlas and Bannister (1980) have shown that the mean spectral coefficient of waters may vary considerably both with the color of the water and the type of algae present. It is probable that the small volumes of phytoplankton present in the clear lakes, together with varying amounts of nonalgal light absorption (background attention), could account for some of the differences observed (Lorenzen 1980, Megard et al. 1980).

TABLE 13. *Selected characteristics of lakes with different levels of measured color (random data set).*

Measured Color Range (units)	Statistic	Color (units)	Secchi Disc (m)	Chlorophyll *a* (μg/l)	pH (units)	Alkalinity (mg/l)	Turbidity (JTUs)	Total P (mg/l)
0-40	No.	365	319	356	364	364	361	363
	Mean	16	2.6	11.8	7.1	40	3.3	0.027
	SD	11	1.5	33.1	0.9	49	5.6	0.044
40-100	No.	159	146	153	159	159	158	159
	Mean	69	1.7	16.5	7.1	55	2.9	0.026
	SD	15	0.9	20.9	0.8	60	2.2	0.033
>100	No.	36	32	32	36	36	33	36
	Mean	146	1.0	12.7	6.8	43	2.5	0.032
	SD	42	0.5	7.4	0.6	63	1.0	0.029

Dissolved Oxygen Conditions

Relationships between dissolved oxygen conditions and other physical and chemical characteristics of lakes have been previously reported (Hutchinson 1938, Deevey 1940, Hasler 1947). Recently, Mathias and Barica (1980) found an inverse relationship between both winter oxygen depletion rate and mean depth and trophic status of Canadian lakes. Welch et al. (1976) demonstrated that in lakes that stratify summer areal chlorophyll *a* and spring areal total phosphorus were strongly correlated with the oxygen depletion rates in the hypolimnion.

Since our data were insufficient for the computation of oxygen depletion rates, an alternative method was devised for evaluating the association of various lake characteristics with summer hypolimnetic dissolved oxygen conditions. By coding all lakes in the total data set according to their "worst case" dissolved oxygen condition, comparisons of the means for the various characteristics associated with "severe" dissolved oxygen condition lakes and stratified lakes were made in order to estimate the association of each of the characteristics with summer dissolved oxygen conditions in the hypolimnion. The stratified lake data group contained lakes with severe oxygen depletion (< 1 mg/l throughout entire hypolimnion), moderate oxygen depletion (< 1 mg/l somewhere in hypolimnion), and lakes with dissolved oxygen concentrations greater than 1 mg/l throughout the hypolimnion; therefore, significant differences noted between all stratified lakes and lakes exhibiting severe dissolved oxygen conditions were deemed valid (Append. A).

Wisconsin lakes exhibiting severe summer hypolimnetic dissolved oxygen conditions generally had slightly below-average water clarity; mean water clarity for all stratified lakes was 2.7 m while the clarity for lakes with severely stressed hypolimnia averaged 2.0 m (Append. A). Eighty-six percent of all stratified lakes with summer water clarity readings less than 2.0 m experienced severe dissolved oxygen stress in their hypolimnia (Table 14). The severely stressed lakes were generally shallower (both mean and maximum depths), had smaller hypolimnetic volumes, and had slightly higher organic nitrogen and total phosphorus concentrations in the epilimnion than all stratified lakes. This is consistent with the findings of other investigators.

Chlorophyll *a* levels in lakes with dissolved oxygen severe stress in the hypolimnion were significantly higher than those for all stratified lakes — 15 μg/l vs 10.5 μg/l (Append. A). Seventy-eight percent of all stratified lakes with chlorophyll *a* concentrations greater than 10 μg/l had severe hypolimnetic dissolved oxygen conditions, while only 40% of the lakes with chlorophyll *a* less than 10 μg/l had the same conditions (Table 14).

Although differences in mean total phosphorus concentrations between lakes with severe dissolved oxygen stress in the hypolimnion and all stratified lakes were not statistically significant (Append. A), progressively higher percentages of stratified lakes showed severe dissolved oxygen conditions at higher phosphorus levels (Table 14). Slightly higher color and total nitrogen levels also were observed in

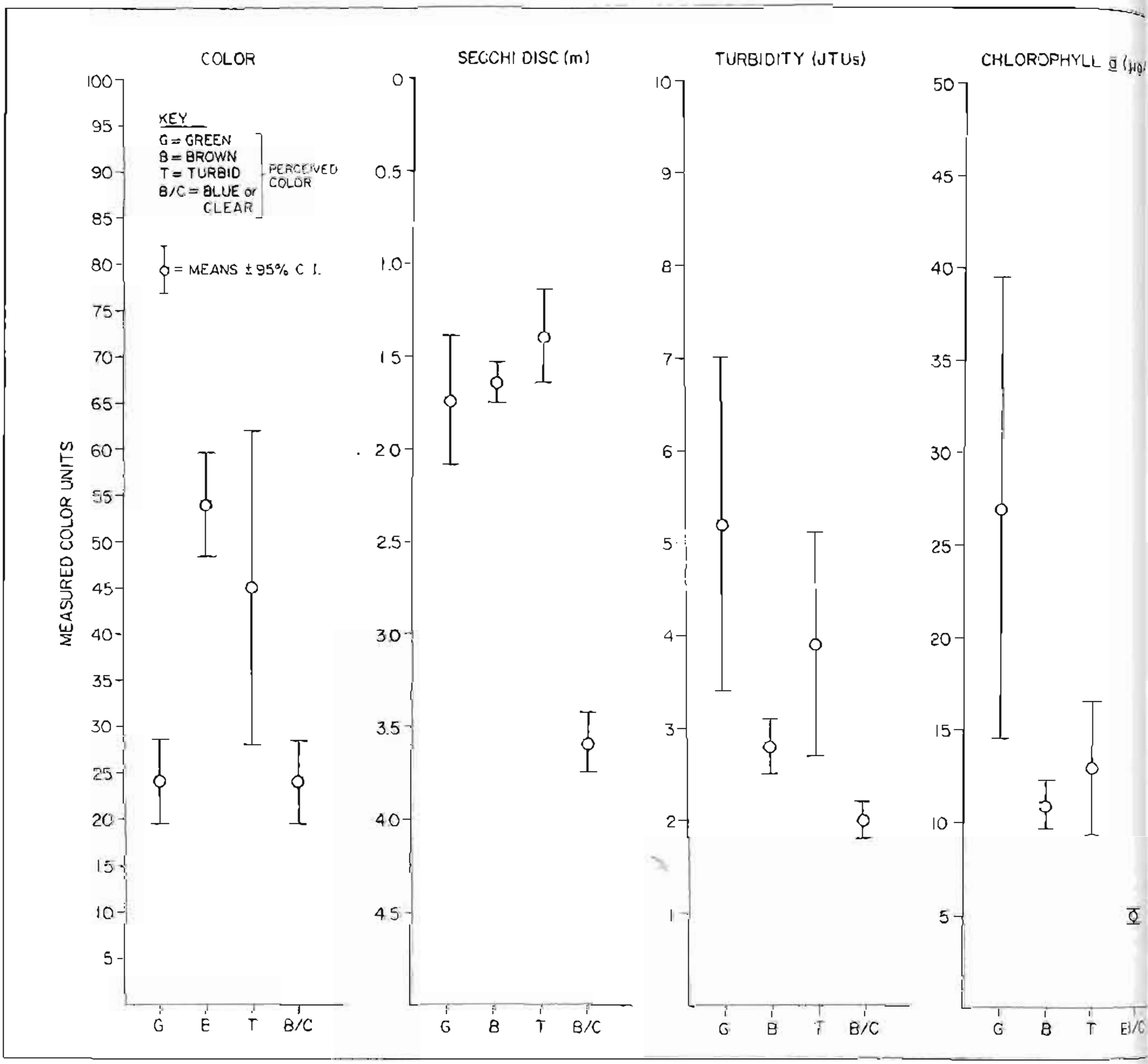


FIGURE 15. *Limnological characteristics of lakes by apparent color type (random data set).*

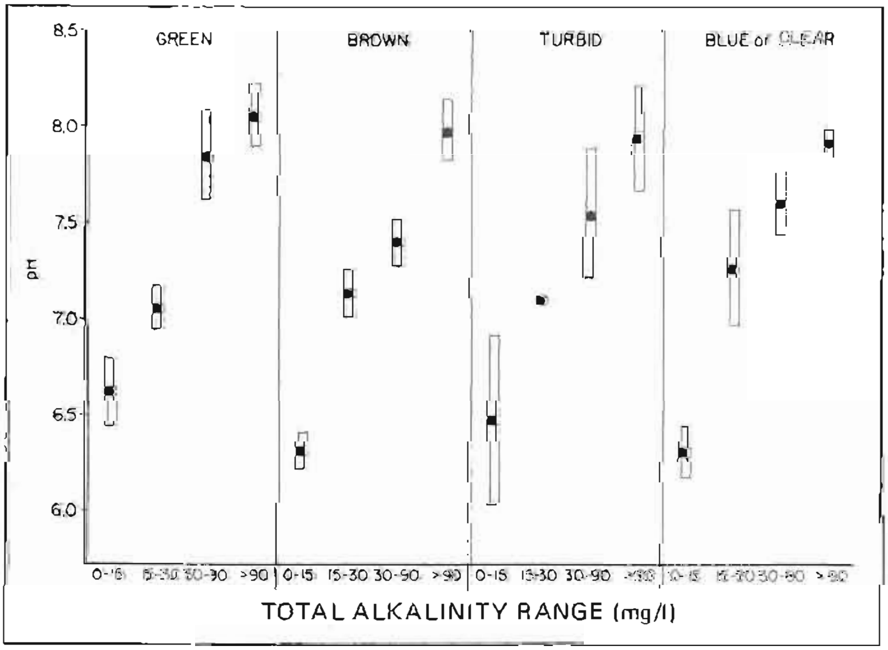


FIGURE 16. *Mean pH (± 95% C.I.) for lakes related to color and alkalinity (random data set).*

lakes with hypolimnetic dissolved oxygen depletion (Append. A). No other significant differences were found between the two groups of lakes.

In order to best describe the associated characteristics of lakes that experience severe winter dissolved oxygen stress conditions, it is first necessary to separate the lakes on the basis of their drainage type.

Summer chlorophyll *a* and total phosphorus (both winter and summer concentrations) were considerably higher in oxygen-stressed lakes than in lakes with no wintertime dissolved oxygen problems evident (Table 15).

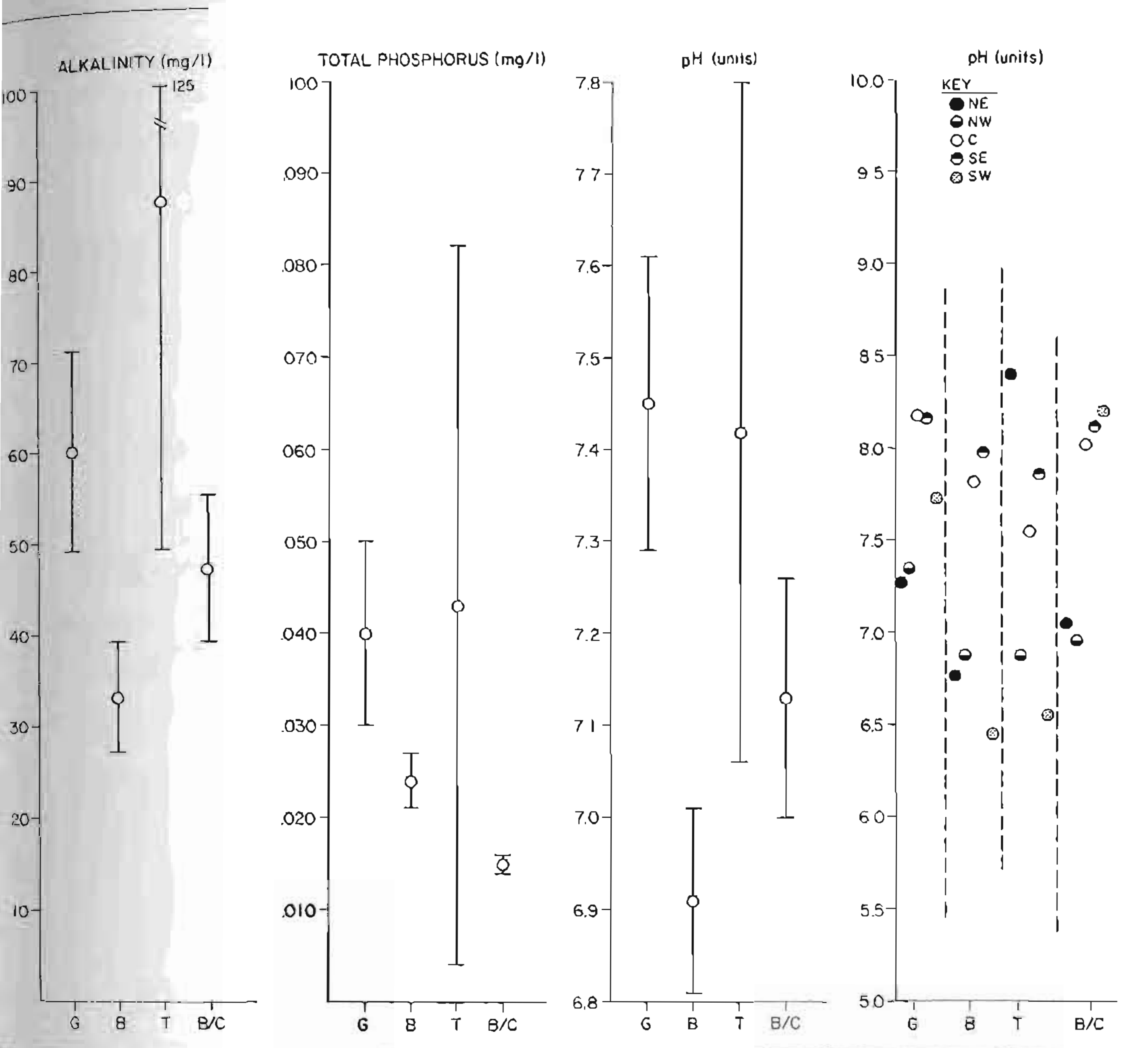


FIGURE 15 (Cont.)

Scidmore (1970) reported that most winterkill lakes in Minnesota had total phosphorus (assume winter) levels greater than 0.05 mg/l and total nitrogen greater than 0.5 mg/l. Our data indicate that winterkill susceptible drainage lakes had considerably higher levels of winter total phosphorus (0.12 mg/l) than seepage lakes (0.07 mg/l), but in both cases the nutrient concentrations were much higher than in lakes not experiencing dissolved oxygen stress. Seventy-nine percent of the lakes with poor winter dissolved oxygen conditions had summer water clarity of less than 1.5 m.

The inorganic-organic phosphorus ratio differs slightly in waters with low dissolved oxygen in winter between drainage and seepage lakes, as does maximum depth (Table 16). Lakes where maximum depths range from 7 to 10 ft are reported by other investigators to be generally susceptible to development of winterkill conditions (Schoenecker 1970, Nickum 1970). Schoenecker (1970) found that average mean depths of winterkill lakes in Nebraska were generally less than 3.5 ft, whereas our data showed Wisconsin lakes with severe winter dissolved oxygen conditions had mean depths of 5.4 ft and 6.0 ft for seepage and drainage types, respectively. However, our mean depth data were significantly biased in that very few lakes in our data set were shallower than 6 ft (maximum depth). It is known that a high percentage of Wisconsin lakes and marshes less than 6 ft deep develop severe dissolved oxygen conditions, and if a greater number of these had been sampled, the average maximum and mean depths of dissolved oxygen-depleted lakes would have been lower.

On the basis of the Wisconsin data, it is evident that poor winter dissolved oxygen conditions are associated with shallow depth, high concentrations of nutrients, and high summer chlorophyll *a* levels.

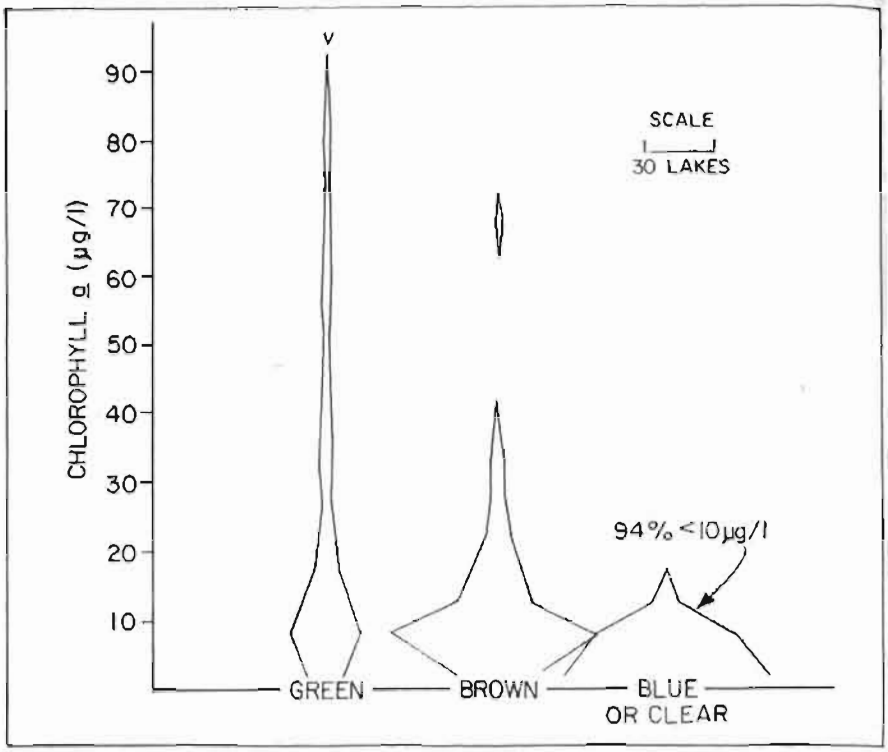


FIGURE 17. *Number of lakes within different chlorophyll a ranges based on perceived water color (random data set).*

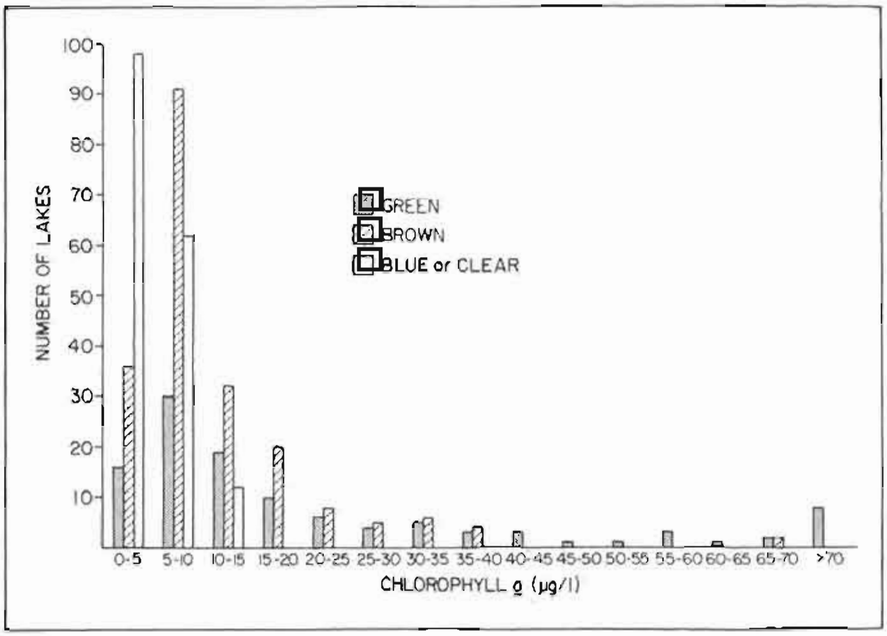


FIGURE 18. *Perceived color groupings according to chlorophyll a concentrations (random data set).*

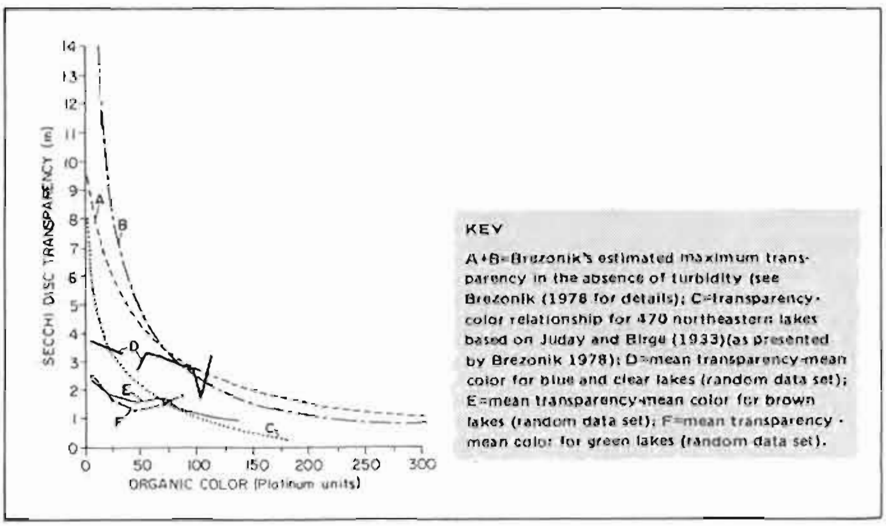


FIGURE 19. *Relationship of transparency to measured color for three groups of Wisconsin lakes with different perceived water color (compared to Brezonik 1978).*

TABLE 14. *Comparative distributions of lakes exhibiting severe dissolved oxygen (D.O.) depletion (summer) and all stratified lakes within a series of selected parameter ranges (quarterly data set).*

Parameter		Stratified Lakes Experiencing Severe D.O. Stress* No. Lakes	Stratified Lakes Experiencing Severe D.O. Stress* Percent of Total	All Stratified Lakes No. Lakes	All Stratified Lakes Percent of Total	Percent of Total Stratified Lakes with Severe D.O. Conditions
Chlorophyll *a* (μg/l, epilimnion)	0-5	23	17	107	38	21
	5-10	59	43	99	36	60
	10-15	15	11	25	9	60
	15-25	17	12	22	8	77
	>25	24	17	25	9	96
Secchi disc (m)	0-1	24	17	26	9	92
	1-2	45	32	54	19	83
	2-3	47	34	87	31	54
	3-4	17	12	66	24	26
	4-5	5	4	30	11	17
	5-6	2	1	11	4	18
	>6	0	0	6	2	0
pH (units, epilimnion)	<5	0	0	0	0	0
	5.0-5.9	10	7	18	6	55
	6.0-6.9	40	28	74	26	54
	>7	91	65	191	67	48
Total Alkalinity (mg/l, epilimnion)	0-15	44	31	86	30	51
	15-30	23	16	38	13	61
	30-90	42	30	86	30	49
	>90	32	23	73	26	44
Total P (μg/l, epilimnion)	< 5	17	12	52	18	33
	5-15	50	35	109	39	46
	15-25	36	26	64	23	56
	>25	38	27	57	20	67

*D.O. = <1 mg/l in entire hypolimnion.

TABLE 16. *Some characteristics of seepage and drainage lakes experiencing severe winter dissolved oxygen conditions (quarterly data set).**

	Lake Type Seepage	Lake Type Drainage
Maximum Depth (ft)	10.8	14.4
Summer Secchi Disc (m)	1.5	1.3
Alkalinity (mg/l)	105	99
Organic N winter (mg/l)	0.91	0.88
Total N winter (mg/l)	1.76	1.80
Inorganic P winter (mg/l)	0.038	0.081
Inorganic P summer (mg/l)	0.059	0.900

*All values represent means.

TABLE 15. *Comparison of characteristics of lakes which show severe dissolved oxygen stress during winter with those that do not (quarterly data set).*

	Parameter		No. D.O. Problem	Moderate Stress* (winter/summer)	Severe Stress* (winter)	Severe Stress Hypolimnion (summer)	Severe Stress (winter/summer)	All Lakes in Data Set
Seepage lakes	Mean depth (ft)	$\bar{x}$	16.8	19.2	5.4	15.7	8.0	16.0
		SD	9.7	8.2	2.08	5.8	0.8	8.6
	Chlorophyll *a* (μg/l)	$\bar{x}$	8.7	11.6	68.0	16.2	34.8	18.5
		SD	8.9	20.5	184.5	16.9	23.7	60.8
	pH	$\bar{x}$	7.24	7.34	7.57	7.46	6.98	7.34
		SD	0.67	0.75	0.63	0.71	0.54	0.70
	Total P (winter) (mg/l)	$\bar{x}$	0.029	0.03	0.066	0.046	0.054	0.038
		SD	0.023	0.02	0.091	0.042	0.021	0.041
	Total P (summer)	$\bar{x}$	0.027	0.03	0.089	0.039	0.058	0.037
		SD	0.025	0.03	0.128	0.029	0.036	0.049
	Drainage basin:lake area	$\bar{x}$	5.72	13.44	5.50	9.57	7.000	8.26
		SD	6.47	29.29	2.71	8.19	2.550	14.32
	Retention time (yrs.)	$\bar{x}$	2.79	2.92	0.89	1.87	0.93	2.38
		SD	1.94	1.77	0.38	1.08	0.09	1.71
	Total lakes no.		8036	17	52	6	191	
	%		41.9	18.9	8.9	27.2	3.1	100.0
Drainage lakes	Mean depth	$\bar{x}$	13.2	13.2	6.04	16.3	6.7	13.4
		SD	13.3	6.9	3.34	7.5	2.8	10.5
	Chorophyll *a*	$\bar{x}$	34.4	27.1	50.8	23.0	119.2	34.9
		SD	52.4	31.3	70.4	25.3	251.7	68.5
	pH	$\bar{x}$	7.54	7.58	7.60	7.69	7.55	7.59
		SD	0.439	0.481	0.369	0.507	0.338	0.462
	Total P (winter)	$\bar{x}$	0.063	0.046	0.124	0.055	0.175	0.066
		SD	0.068	0.046	0.237	0.051	0.259	0.100
	Total P (summer)	$\bar{x}$	0.069	0.061	0.163	0.047	0.215	0.073
		SD	0.074	0.069	0.267	0.039	0.237	0.109
	Drainage basin:lake area	$\bar{x}$	395.46	148.18	64.38	172.13	114.80	254.56
		SD	836.48	285.91	90.32	835.57	227.79	729.93
	Retention time	$\bar{x}$	1.16	1.03	0.04	1.13	0.57	1.06
		SD	2.19	1.28	0.38	1.33	0.46	1.72
	Total lakes no.		151	59	26	98	10	344
	%		43.9	17.2	7.56	28.5	2.9	100.0

* Moderate = <1 mg/l somewhere in hypolimnion; Severe = 1 mg/l in entire hypolimnion.

REGIONAL DESCRIPTIONS

There is a considerable amount of overlap in the ranges of many water quality values found in lakes throughout the state, and many of these values fail to show any distinctive geographical gradient. However, by clustering lakes within designated geographic regions (Fig. 3), calculations of regional means can be made and compared for statistical significance. Average characteristics of lakes within these regions are to a considerable degree dependent upon the physical and chemical characteristics of the region — the watershed geology, soils, land use, topography, climate and the lake basin morphometry. The number of lakes of various lake types (e.g., seepage vs drainage) found within each region had an important influence on the analysis of differences between regions.

Data on regional comparisons of various parameters, upon which the following discussions are based, may be found in Tables 17 and 18 and Figures 20-24.

Northeast Region

Most of the state's lakes (80%) larger than 25 acres and more than 5 ft deep are located in the northern regions of the state. The Northeast Region has 37% of the lakes in the state, of which 93% are natural in origin — a higher percentage than in any other region.

There are almost equal numbers of drainage and seepage lakes and mixed and stratified lakes in this region. A high percentage of the lakes are less than 250 acres in size but a large number of the state's lakes larger than 250 acres are also found in the region. Mean surface acreage of Northeast Region lakes greater than 25 acres and more than 5 ft deep is 198 acres. The Northeast lakes sampled in the random survey were generally deeper than the statewide average; mean depth was 15 ft and maximum depth averaged 28 ft. Water clarity (2.7 m) was above average and ranked best among the five regions. The relatively high mean water color (46 units) and low pH (6.9 units) for the region is probably indicative of the large number of brown-stained lakes (39%) in the region. Sixty-seven percent of the state's lakes with pH below 6.0 were located in the Northeast Region, and 54% of the region's lakes had alkalinities less than 30 mg/l. Northeast Region lakes generally had low levels of calcium, magnesium, chlorides, turbidity, nutrients and chlorophyll *a*. Sixty-three percent of the region's lakes had total phosphorus levels less than 0.015 mg/l, and 72% had chlorophyll *a* levels less than 10 μg/l. The regional mean for chlorophyll *a* was low, 9 μg/l.

Northwest Region

In this region there is a slightly higher proportion of seepage lakes (56%) than drainage lakes, while mixed and stratified lakes are nearly equally divided. Northwest Region lakes in the random sample were mostly less than 100 acres in size and were slightly smaller (avg. size = 165 acres) than Northeast Region lakes. While 47% of the region's lakes greater than 25 acres and greater than 5 ft deep were described as brown in color, mean measured water color (30 units) was the lowest of the five designated regions. Northwest Region lakes were generally low in calcium, magnesium and chlorides and had near state average turbidities and total nitrogen levels. Sixty-two percent of the region's lakes had alkalinities less than 30 mg/l and 40% had alkalinities less than 15 mg/l. Mean alkalinity was 27 mg/l. The mean total phosphorus concentration (0.028 mg/l) for Northwest Region lakes was about the same as the statewide average for randomly sampled lakes. While the majority of the Northwest Region lakes had low total phosphorus levels (47% less than 0.015 mg/l), 44% of the state's lakes with phosphorus concentrations exceeding 0.035 mg/l were found here. Most of these were shallow eutrophic lakes located in Polk, St. Croix and Barron counties, which are not typical of the remainder of lakes in the Northwest Region. Whether or not these counties should be included in this region or some other is a valid question that was considered in regional delineations. Twenty-two percent of the region's lakes had chlorophyll *a* levels above 15 μg/l, representing 41% of the state total in that category. Water clarity varied considerably; the mean of 2.1 m was slightly less than the statewide average. The majority of lakes had water clarity ranging between 1 and 3 m.

Central Region

Most of the Central Region lakes are located in the central plain geographic province which is underlain with sandstone formations. The glacial deposits and soils in the region reflect this in that they generally contain a considerable amount of sand. Lakes in the central region are clustered in specific locations and in a large part of the region there is a scarcity of lakes due to the nature of the underlying soils and bedrock (Martin 1965).

Seventy-three percent of the sampled lakes in the region were of natural origin, divided equally between drainage and seepage and mixed and stratified types. The majority (77%) of the lakes were smaller than 100 acres in size. Water quality was generally very good in Central Region lakes; water clarity was better than the statewide average. Mean total nitrogen (0.72 mg/l) was lower than in other regions except the Northeast. The total phosphorus content of the region's lakes (mean 0.020 mg/l and median 0.012 mg/l) was very low; total phosphorus concentrations in 70% of the lakes were below 0.015 mg/l. Mean chlorophyll *a* concentration was lowest of any of the five regions with 77% of the lakes having less than 10 μg/l. Turbidity was slightly below the statewide average (2.6 JTUs). Water color was somewhat above the statewide average, but most Central Region lakes were perceived as clear by sampling teams and only 10% appeared green at the time of field sampling.

The Central Region had better water quality than might be expected for a group of lakes with such high alkalinities; alkalinities closely resembled those of the southern regions, but nitrogen, phosphorus, chloride, turbidity, chlorophyll *a* and water clarity means were similar to those found in the northern regions. The reasons for this apparent disparity are uncertain, but may be related to the generally small lake size (mean 84 acres), relatively small watersheds, and long water retention times of the Central Region lakes. These factors result in lower nutrient loading and lower levels of nitrogen and phosphorus in the lakes. If, because of high alkalinity, precipitation of marl was removing total phosphorus from the surface water, it might be expected that the Central Region lakes would have significantly higher nitrogen:phosphorus ratios than lakes in other regions, but this does not appear to be the case. Mean nitrogen:phosphorus ratios were slightly higher, but not significantly. It is possible that input of phosphorus was compensating for losses, but this would be expected to be accompanied by higher total nitrogen levels, which were not observed. Whatever the reason, the Central Region lakes form a rather distinct lake group with average characteristics significantly different than those of lakes in other regions.

Southeast Region

Limestone and dolomite deposits make up much of the underlying bedrock of lake watersheds in the Southeast Region. As a result, high alkalinity (173 mg/l) and calcium and magnesium levels (36 and 32 mg/l, respectively) were characteristic of lakes in the region. More than 95% of the region's lakes had alkalinities in excess of 90 mg/l, and high pH (mean = 8.2 units) accompanied the high alkalinities.

Lakes sampled in the Southeast Region exhibited great variability in their water quality characteristics. Sixty-one percent of the region's sampled lakes were drainage lakes, while 77% were of natural origin. Mixed lakes outnumbered stratified lakes, and mean depths (10 ft) were slightly less than the state average. There were more large lakes (greater than 1000 acres) in the Southeast Region sample than in other regions; as a result, the average lake size was 582 acres.

Water quality and clarity in Southeast Region lakes were generally below average. Average water clarity was only 1.6 m, which is only fair, and may be attributed to a combination of relatively high turbidity (6.6 JTUs), above-average mean color (46 units), and high chlorophyll *a* levels (mean = 43 µg/l). Thirty-four percent of the Southeast Region lakes had chlorophyll *a* levels above 15 µg/l, while median chlorophyll *a* was 10 µg/l. Fifty-four percent of the lakes had total phosphorus levels above 0.025 mg/l, resulting in a very high phosphorus concentration mean of 0.079 mg/l. Mean total nitrogen (1.42 mg/l) was higher than that found in any other region. The generally high trophic status of the Southeast Region lakes was quite apparent as 47% of the region's lakes were identified as green in appearance.

TABLE 17. *Various physical, chemical and biological characteristics of regional lakes (random data set).**

	Chlorophyll *a* (µg/l)					Secchi Disc (m)				
	0-5	5-10	10-15	15-25	>25	0-1	1-2	2-3	3-4	>4
Northeast	81	90	27	23	15	38	63	54	44	44
Northwest	78	99	37	29	30	48	74	73	34	21
Central	15	19	4	4	1	2	11	15	7	3
Southeast	13	18	8	6	14	18	17	14	5	0
Southwest	0	4	5	6	15	17	9	2	0	0

	Total Phosphorus (mg/l)					Color (units)			
	<.005	.005-.015	.015-.025	.025-.035	>.035	0-10	10-40	40-100	>100
Northeast	53	99	50	22	18	40	71	77	19
Northwest	33	99	66	32	52	71	116	66	13
Central	14	17	4	3	6	7	13	13	3
Southeast	4	13	9	11	24	5	12	11	3
Southwest	0	2	7	3	18	1	8	5	6

	Alkalinity (mg/l)				pH (units)			
	0-15	15-30	30-90	>90	<5	5-6	6-7	>7
Northeast	94	38	78	33	7	33	63	140
Northwest	114	62	99	7	0	17	112	153
Central	1	1	6	36	0	0	1	43
Southeast	0	0	3	58	0	0	0	61
Southwest	9	3	8	10	0	3	7	20

	Perceived Color			
	Green	Brown	Turbid	Blue or Clear
Northeast	23	61	2	72
Northwest	59	124	5	75
Central	4	11	7	18
Southeast	18	6	5	9
Southwest	9	9	2	3

*All values represent number of lakes.

Southwest Region

Southwest Region lakes are best described as shallow, eutrophic drainage lakes and impoundments. Only 10% of the lakes sampled were of natural origin and only 3% were seepage lakes. The scarcity of natural lakes is attributed to the topography and geological history of the region, much of it included in the Driftless Area. The relatively shallow mean depths (avg. = 6.5 ft) of the region's lakes were reflected in the fact that 87% of the lakes were thermally mixed. Average lake size was quite large (mean = 915 acres). Lakes of the Southwest Region have generally poor water quality, which is directly related to the fact that most of the lakes are impoundments. High levels of color (mean = 68 units) and chlorophyll *a* (mean = 32 µg/l, median = 25 µg/l) contributed to the poor water clarity (mean = 1.0 m). Fifty percent of the Southwest Region lakes had chlorophyll *a* concentrations exceeding 25 µg/l, while only 7% had water clarity readings over 2.0 m. Total phosphorus concentrations exceeding 0.025 mg/l were found in 70% of the region's lakes.

Regional Summary

Water quality varies considerably between and within the five regions, as shown by the wide ranges in values and confidence intervals about the means. Generally, lowest levels of nutrients and the overall best water quality and clarity are coincidental, or occur together. There is a higher percentage of good water quality lakes in the northern region of the state than in the southern region, but each region has some lakes with poor water quality.

Differences in lake types, and the numerical distribution of the different types of lakes within geographical regions contributed to the difficulties involved in interpreting the data. The many factors which combine to determine water quality characteristics of lakes cannot be completely separated. For example, mean levels of total phosphorus, alkalinity and chlorophyll *a* vary considerably, depending on the number of seepage or drainage lakes within a region, since seepage lakes have lower alkalinities, total phosphorus and chlorophyll *a* concentrations. While not shown, this same principle applies to stratified lakes vs mixed lakes. Therefore, when making regional comparisons of lakes it is important to take into account the proportion of various lake types found within each region. While it would be preferable to make comparisons between lakes with nearly identical physical characteristics, such ideal situations rarely exist. Because of the numerous subtle differences in the water quality of lakes in the five described regions, it is difficult and not justifiable to make further comparisons, unless they are made between lakes with nearly identical physical characteristics.

TABLE 18. *Summary of characteristics of Wisconsin lakes by region (random data set).*

Parameter	Area (acres)	Mean Depth (ft)	Max. Depth (ft)	Color (units)	Secchi Disc (m)	Chlorophyll *a* (µg/l)	Chlorides (mg/l)	Calcium (mg/l)	Magnesium (mg/l)	pH (units)	Alkalinity (mg/l)	Turbidity (JTUs)	Organic N (mg/l)	Total N (mg/l)	Inorganic P (mg/l)	Total P (mg/l)
NORTHEAST																
No. lakes	243	46	243	207	225	236	189	189	189	243	243	233	243	243	242	242
Mean	198	15.4	28	46	2.7	9.3	2	10	5	6.9	37	2.0	0.52	0.66	0.006	0.019
Standard deviation	297	7.1	18	46	1.6	8.3	2	8	6	0.9	40	1.5	0.33	0.39	0.007	0.013
Median	82	14.8	23	33	2.4	6.7	2	7	2	7.1	22	1.5	0.42	0.55	0.004	0.016
Minimum	25	6.0	6	1	0.5	1.0	1	1	1	4.3	1	0.5	0.10	0.17	0.001	0.003
Maximum	1,918	34.4	98	320	9.5	50.2	11	51	26	8.9	224	10.0	2.17	2.52	0.048	0.092
NORTHWEST																
No. lakes	282	125	282	266	250	273	282	282	282	282	282	277	282	282	281	282
Mean	165	11.4	24	30	2.1	12.4	2	7	3	7.0	27	3.4	0.62	0.89	0.011	0.028
Standard deviation	369	6.7	17	31	1.2	14.5	2	6	3	0.7	25	3.7	0.32	0.43	0.015	0.026
Median	67	9.6	18	20	2.0	7.6	1	5	1	7.0	18	2.4	0.55	0.79	0.006	0.020
Minimum	25	2.7	5	2	0.3	0.5	1	1	1	5.4	1	0.8	0.12	0.14	0.001	0.003
Maximum	3,227	40.0	105	140	6.0	100.7	19	35	16	9.6	133	34.0	2.28	2.84	0.132	0.173
CENTRAL																
No. lakes	44	25	44	36	38	43	44	44	44	44	44	44	43	43	44	44
Mean	84	13.1	30	42	2.4	7.5	4	24	20	7.9	122	2.6	0.48	0.72	0.007	0.020
Standard deviation	77	10.0	22	36	1.2	4.8	2	8	7	0.4	40	1.6	0.23	0.31	0.011	0.021
Median	44	10.2	26	28	2.1	6.2	4	22	21	7.9	124	2.0	0.41	0.69	0.004	0.012
Minimum	25	2.0	6	1	0.7	1.9	1	3	1	6.7	12	0.7	0.19	0.30	0.001	0.005
Maximum	272	36.9	95	130	7.3	25.0	10	42	31	8.9	190	8.4	1.16	1.56	0.058	0.110
SOUTHEAST																
No. lakes	61	23	61	31	54	61	61	59	59	61	61	61	61	61	61	61
Mean	582	10.2	25	46	1.5	43.3	19	36	32	8.1	173	6.7	0.94	1.43	0.048	0.079
Standard deviation	1,880	8.6	20	40	0.9	116.0	14	16	9	0.5	55	11.2	0.49	1.10	0.104	0.136
Median	89	6.6	16	30	1.4	9.9	16	31	34	8.0	160	3.0	0.77	1.18	0.015	0.030
Minimum	25	2.4	5	3	0.1	1.8	1	8	2	7.1	51	1.1	0.31	0.43	0.001	0.008
Maximum	10,460	40.0	85	160	3.7	706.1	57	71	49	9.4	290	72.0	2.77	6.50	0.570	0.720
SOUTHWEST																
No. lakes	30	15	29	20	28	30	30	30	30	30	30	30	30	30	30	30
Mean	915	8.1	16	68	1.0	32.0	7	16	9	7.2	67	3.6	0.54	1.19	0.036	0.067
Standard deviation	4,028	4.3	11	60	0.5	27.4	4	13	10	0.8	65	2.9	0.21	0.69	0.033	0.052
Median	98	6.2	10	60	0.9	24.8	6	12	6	7.2	42	2.7	0.52	0.92	0.022	0.050
Minimum	25	3.2	6	7	0.5	5.8	1	1	1	5.7	2	1.1	0.19	0.48	0.004	0.013
Maximum	22,218	15.9	44	220	2.7	122.3	21	49	33	9.2	202	14.0	1.02	3.49	0.126	0.224

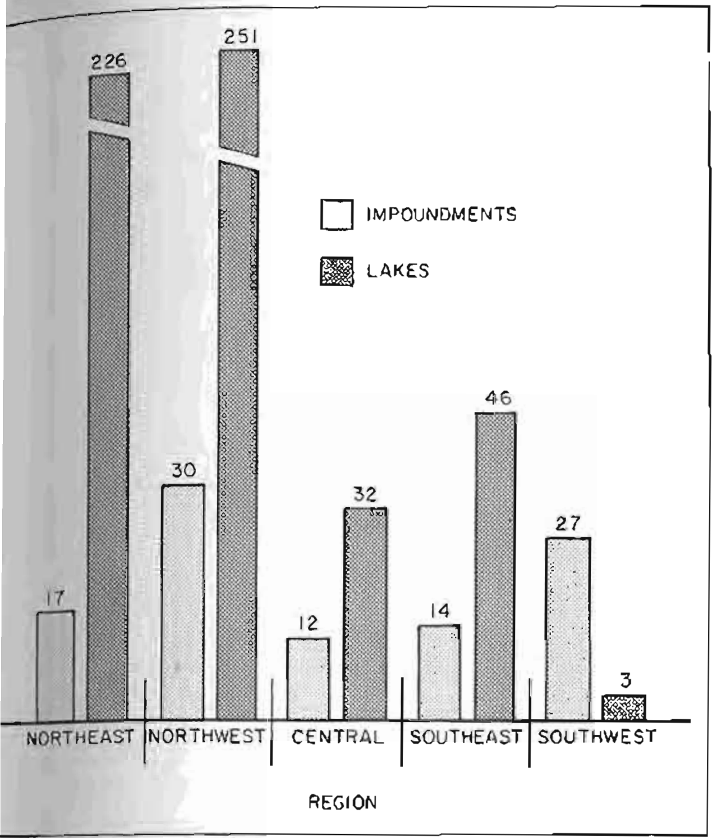


FIGURE 20. *Distribution of natural lakes and impoundments by region (random data set).*

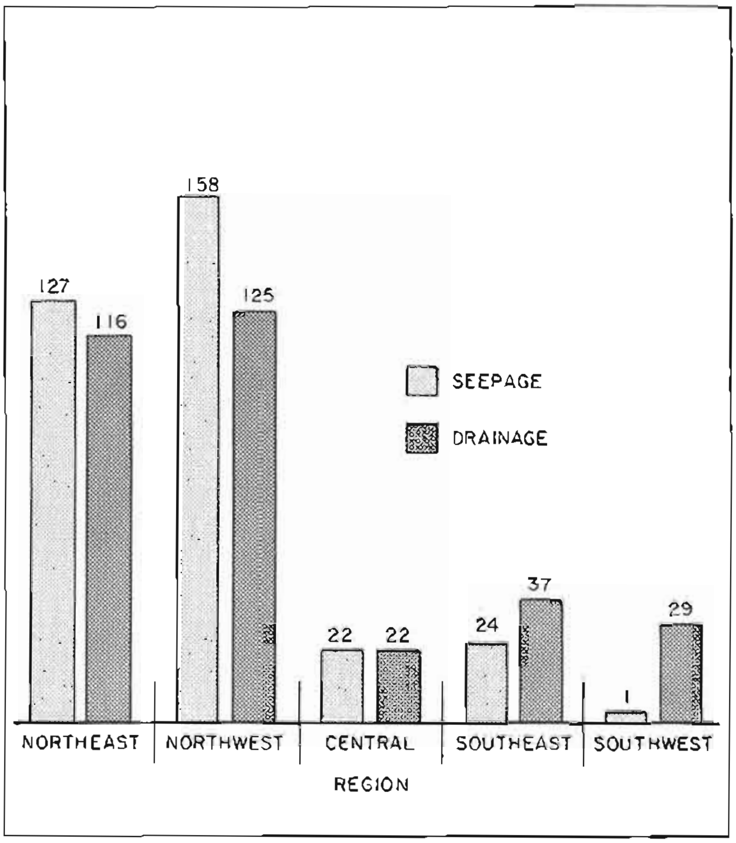


FIGURE 21. *Distribution of seepage and drainage lakes by region (random data set).*

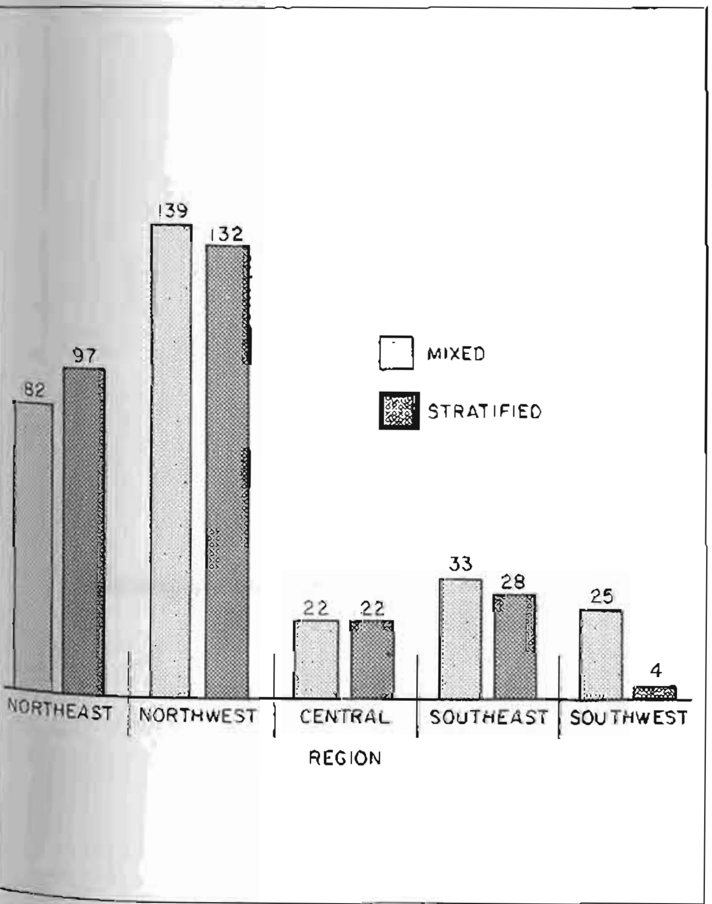


FIGURE 22. *Distribution of mixed and stratified lakes by region (random data set).*

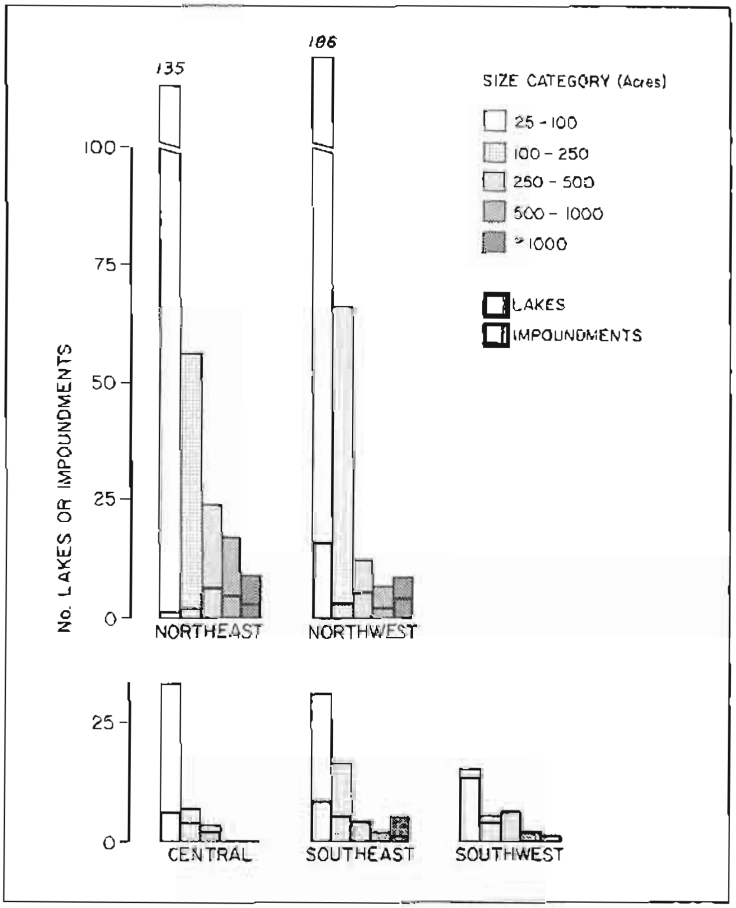


FIGURE 23. *Distribution of lakes and impoundments within the random data set by region and size.*

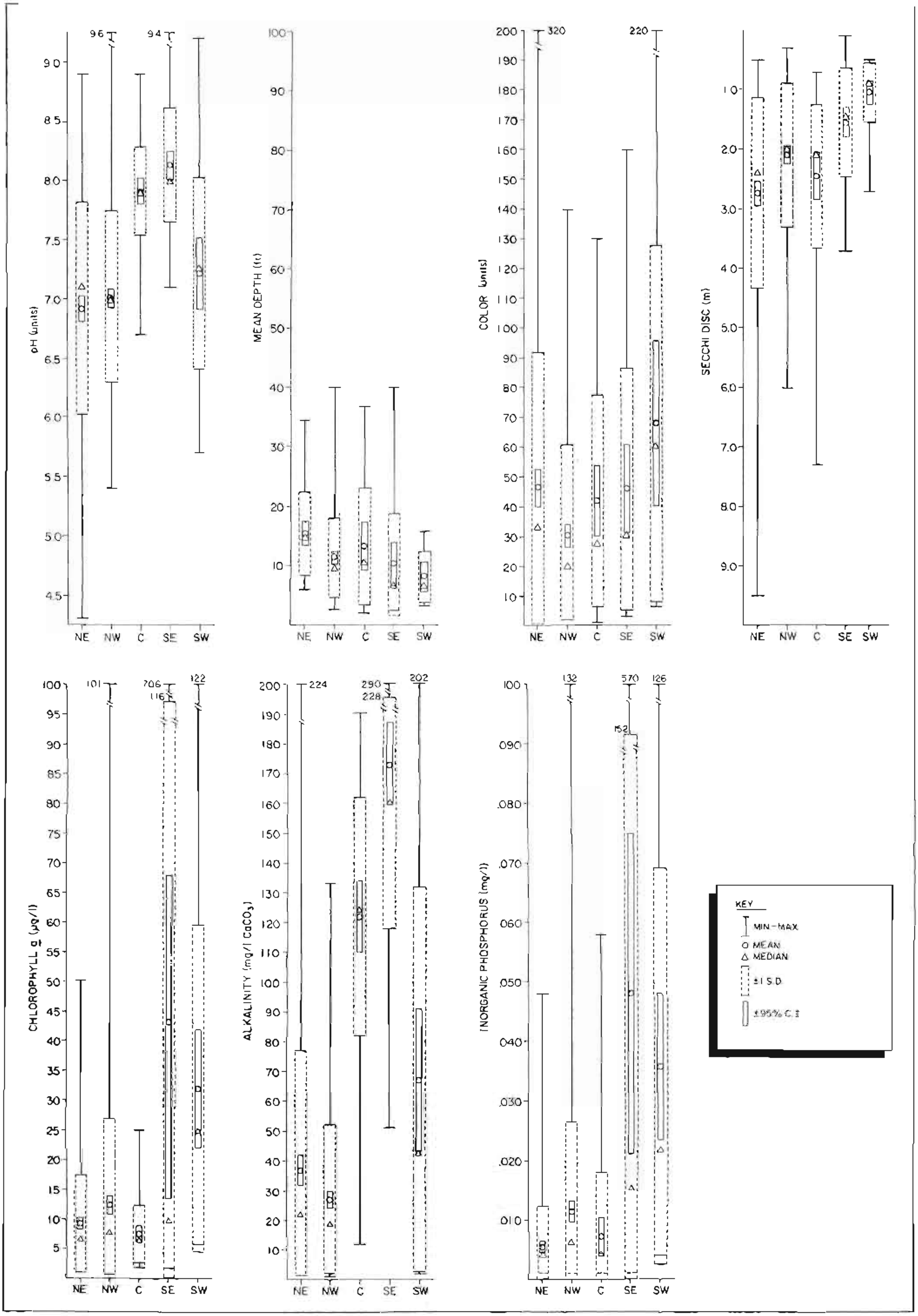


FIGURE 24. *Characteristics of Wisconsin lakes by region (random data set).*

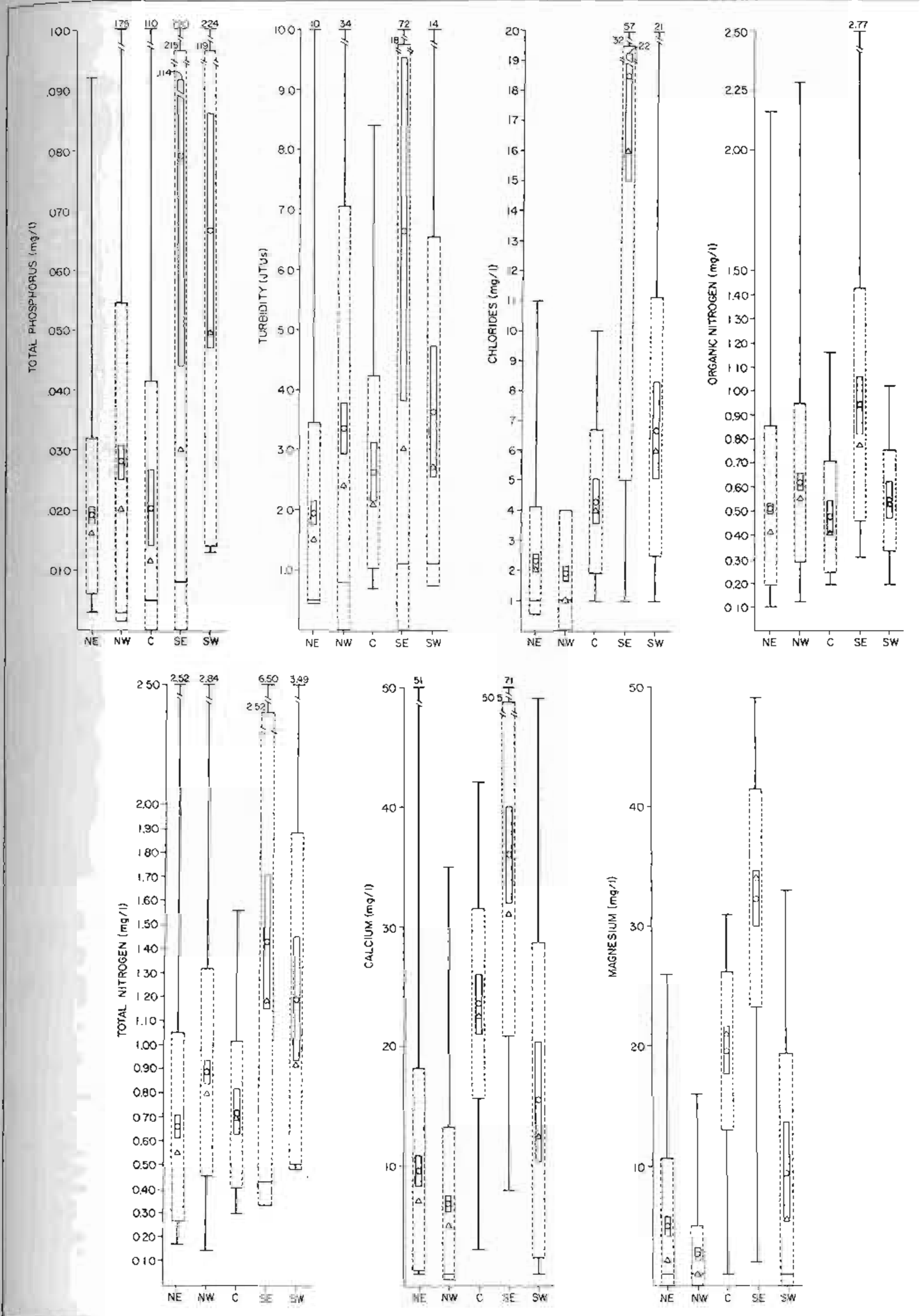


FIGURE 24 (Cont.)

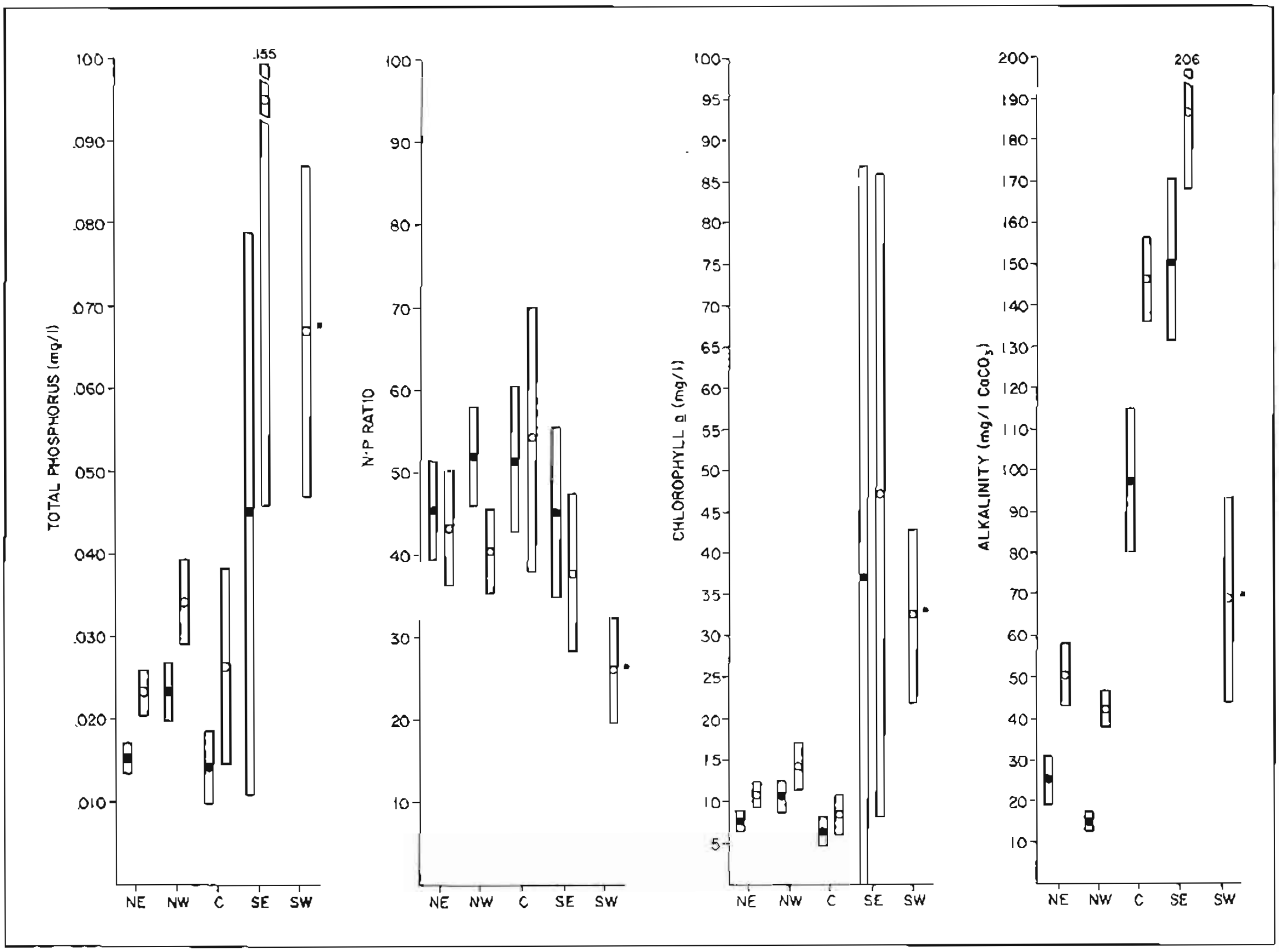


FIGURE 24 (Cont.)

● Seepage ○ Drainage

* Data representing 1 seepage lake in this region is not shown.

Aquatic plants are considered to be very beneficial in some Wisconsin lakes, but they are a nuisance problem in others.

Wisconsin's "clear" lakes, found mostly in the northern and central parts of the state, are popular for recreational activities.

TROPHIC CLASSIFICATION AND INFLUENCING FACTORS

Lake water quality data are generally collected with three major objectives in mind: (1) to assess existing water quality conditions for immediate management purposes such as manipulation of fish populations or pollution control programs, (2) to document existing conditions as a basis for assessing changes in water quality with time. e.g., monitoring expected improvements in water quality following construction of wastewater treatment facilities or determining the long-term impact of acid deposition on lake ecosystems, and (3) to gain a better understanding of the factors and interrelationships which affect water quality in lakes. The data collected during this 14-year study, which involved 1,140 different Wisconsin lakes, has already helped meet these objectives and will continue to act as a data base for future reference.

The assessment of water quality of lakes is based on comparisons of various characteristics which are considered to be indicative of "good" or "poor" conditions. Therefore, the actual assessment of the water quality of any particular lake is dependent on (1) the conception of the individual making the assessment, (2) the parameter selected for making the assessment, including its natural variability, (3) the accuracy and precision inherent in measuring the selected parameter, and (4) the reliability of parameter values used in making qualitative delineations.

Individuals have different perceptions as to what constitutes "good" water quality, as indicated by the number of different parameters and parameter values used to judge water quality conditions (Shapiro 1975). "Ideal" water quality depends upon the point of view of the individual; for example, extensive beds of macrophytes may be a real nuisance to the water-skier, but to the fishermen they provide shelter and habitat for fish. Therefore, different individuals give different values to various parameters in making an assessment of water quality.

Water quality indicators that are chosen for making evaluations may be stable or may experience great daily, seasonal or annual fluctuations; the significance of this fact can not be overestimated nor overlooked. Sampling and analytical variability or errors may also compound the problem of accurately assessing water quality (e.g., see Tyler 1968). However, possibly most important of all is the translation of quantitative measurements to a qualitative statement or assessment. This function of categorization or classification is entirely dependent upon the selection of criteria or "standards" used for delineation. Again, the selection or determination of these criteria generally rests with the individual making the assessment, since literature values suggested as classification boundaries vary considerably for individual parameters. McClelland and Deininger (1981) summed up the situation by stating that "there is still no generally accepted index of water quality for lakes."

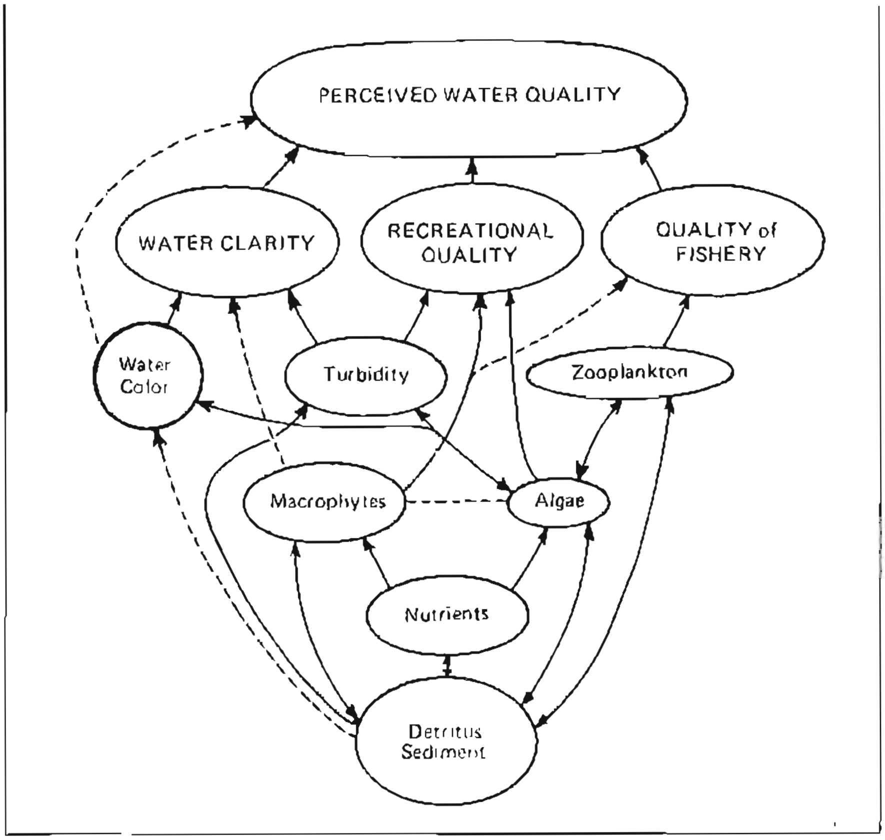


FIGURE 25. *Conceptualization of some of the major in-lake factors affecting perceived water quality. (—) Direct influence, (---) indirect influence.*

Numerous classification systems have been developed for assessing water quality of lakes. The simplest lake classification systems are based on a single differentiating characteristic, such as drainage type, stratification status, dominant algae species, fishery type, etc. An alternative classification form is based on ranking lakes along a continuous parameter (e.g., productivity, biomass, chlorophyll *a*, phosphorus concentration, total alkalinity, etc.) and division of the parameter scale to provide a varying number of classes. This form of classification is typified by the many parameters used to rank lakes according to their trophic status (oligotrophic-mesotrophic-eutrophic). The majority of these trophic scales are based on some relationship to the lake's biological productivity; hence, low values are related to low productivity (oligotrophic) and high values are related to high productivity (eutrophic). Various modifications and transformations of data have been made in an attempt to develop an acceptable and rational trophic classification system (Carlson 1977).

A third type of classification system includes combinations of parameters, or multiparameter classifications, such as Ryder's (1965) Morphoedaphic Index, Brezonik and Shannon's (1971) Multivariate Trophic State Index, Michalski and Conroy's (1972) Lake Evaluation Index, and Uttormark and Wall's (1975*b*) Lake Condition Index.

In addition, various hierarchial-type systems based on methods such as principal component analysis, cluster analysis, and complex ordinate-community structure analysis have been proposed. To be certain, a great number of lake classification systems have been developed, and excellent reviews of this topic are available in Sheldon (1972), Shapiro (1975), Ott (1977), and McClelland and Deininger (1981).

Sheldon (1972) presented excellent arguments against the development of universal classification systems. While splitting or lumping any mass of data is possible, quality of the resulting classification system depends largely upon the purpose for which it was developed (Schneider 1975, Shapiro 1975). The public's perception of water quality, which varies considerably depending on individual preferences, is primarily dependent on water clarity and color, fish production, and extent of macrophyte and algae growth (Fig. 25). Unfortunately, water quality and trophic status are all too often equated without considering the dynamic nature of the water quality characteristic being observed; lakes classified as oligotrophic according to one parameter may in some cases be classified as eutrophic on the basis of some other characteristic.

We will summarize those water quality characteristics of Wisconsin's lakes most often used to rank or classify lakes according to trophic condition, and will discuss relationships between various trophic indicators and the factors affecting these relationships. In addition, a general classification system using apparent water quality characteristics for Wisconsin lakes is proposed based on the results of this study (Table 21).

TROPHIC CLASSIFICATION

Numerous indicators have been used to calculate a lake's trophic state (Vollenweider 1968, Shapiro 1975, Bartsch and Gakstatter 1978, McClelland and Deininger 1981), but most indices rely on water clarity, chlorophyll *a* concentrations, total phosphorus concentrations, or combinations of these three. Carlson (1977) contributed to the popularity of Secchi disc, chlorophyll *a* and total phosphorus as key indicators by developing a numerical scale, the Trophic State Index, based on the interrelationships between them. However, all trophic classification systems have inherent weaknesses, and none should be considered as the complete answer to the problem of classifying lakes.

Water Clarity

The U.S. Environmental Protection Agency has proposed Secchi disc readings of 2.0 and 3.7 m as dividing lines separating lakes according to their trophic state (1975), but these values have not yet become widely accepted. While the values suggested by the National Eutropication Survey (NES) study are undoubtedly as well-founded as any, it is recognized that many factors other than water clarity influence judgment of trophic status or, more correctly, factors other than trophic status affect water clarity. The NES criteria can be applied to Wisconsin lakes; however, based on our data and experiences, water clarity for Wisconsin lakes can be categorized in more descriptive terms as follows:

6 m = Excellent (2%)*
3-6 m = Very Good (24%)
2-3 m = Good (27%)
1 1/2-2 m = Fair } (29%)
1-1 1/2 m = Poor }
<1 m = Very Poor (29%)

*Percent of lakes in random sample.

The mean and median Carlson Trophic State Index (TSI) for all randomly sampled Wisconsin lakes were 48 and 50, respectively (Fig. 26). These Carlson TSI's for Wisconsin lakes correspond to "fair" to "good" water clarity categories under our classification system. Categorization of statewide mean and median water clarity, however, has little significance because of the great variability in water clarity of individual lakes.

Impoundments, drainage lakes and mixed lakes generally have poorer water clarity than natural, seepage and stratified lakes (Fig. 27) and therefore would be ranked lower on the scale. The poorer water clarities of the former lake groups are undoubtedly associated with greater nutrient loading and internal recycling rates, and generally shallower depths (Fig. 28). Among the stratified lakes, lakes experiencing low D.O. conditions in their hypolimnia have generally poorer water clarity than those which did not (Fig. 27).

The Central Region had the lowest percentage of poor water clarity lakes (representing only 2% of the state total), while the Southwest Region had the highest percentage (Fig. 27). Fifty-four percent of the state's lakes with very good clarity (> 3 m) were found in the Northwest Region. While these comparisons could be misleading due to the wide disparity of lake types found within and between the five lake regions, regional differences in water clarity are nevertheless important for lake management. Perceptions of water quality are often based solely on water clarity, which may vary considerably depending on the range of experiences of the viewer. For example, individuals residing in the Southwest Region, who may not have had exposure to lakes of other regions with excellent water clarity, may characterize their lakes with water clarity of 2-3 m as having excellent water clarity. On the other hand, someone from the Northeast Region may consider these same lakes as having poor water clarity. These differences in perspective are a very important management consideration regardless of the water quality parameter chosen for evaluating lake quality, and they must be considered in the development of any broad statewide lake management program.

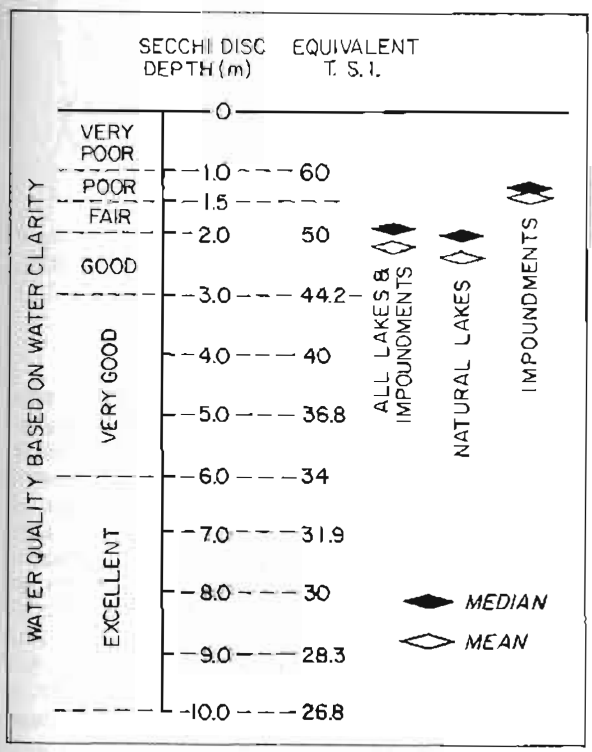


FIGURE 26. *Water clarity for 595 randomly sampled Wisconsin lakes and its relationship to Carlson's (1977) Trophic State Index (TSI).*

Biological Production

The traditional distinctions used in trophic classification of lakes are based on principles of biological productivity (Lindeman 1942). The rate of production of organic matter is not necessarily positively correlated to the total biomass. Very eutrophic lakes can have low productivity rates on an areal basis while having high standing crop biomass levels and, conversely, oligotrophic lakes can have high relative pro-

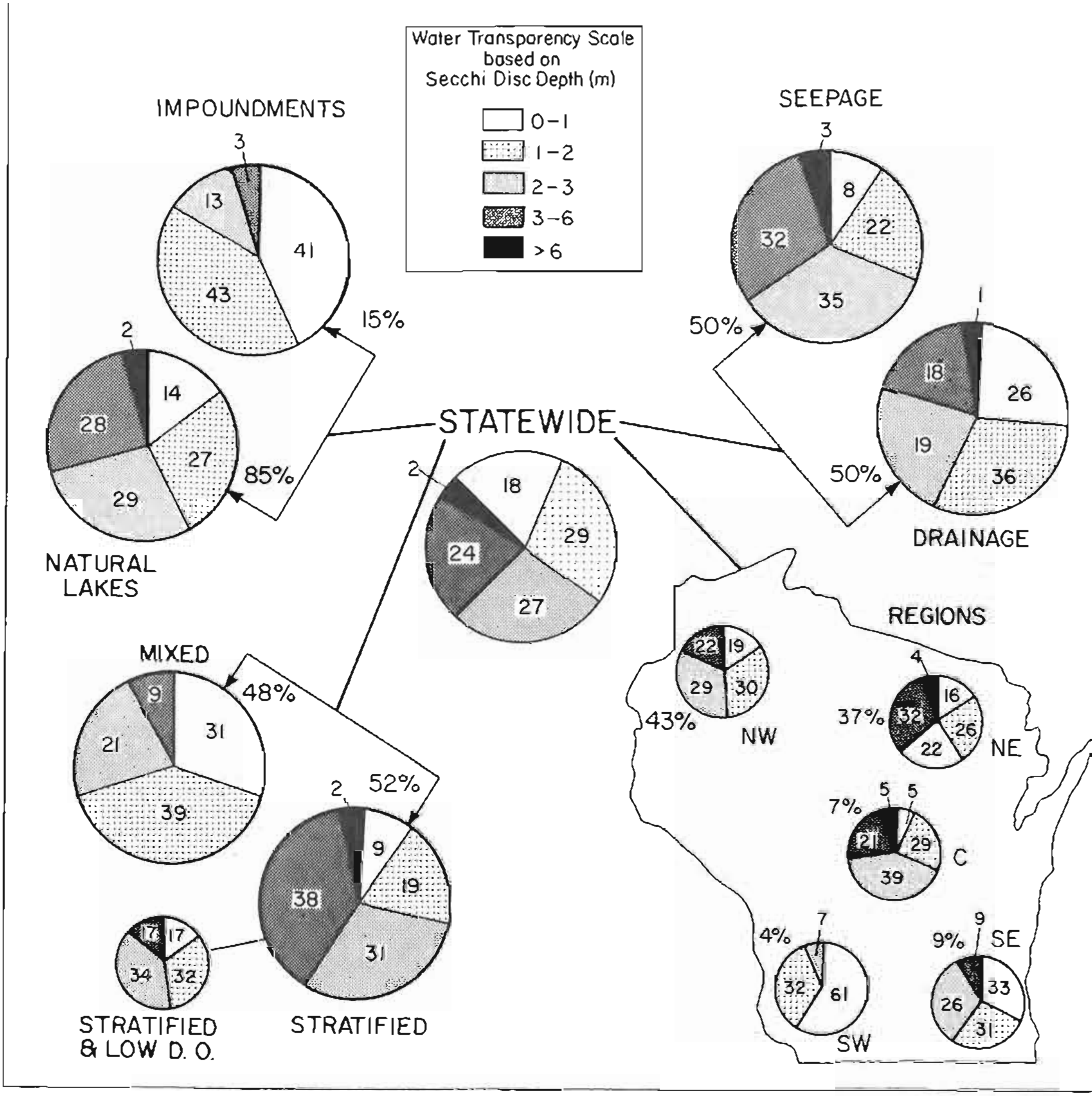


FIGURE 27. *Percent distribution of different classes of lakes based on water transparency.*

ductivity rates and low standing crop biomass. However, biomass, whether in terms of fish, insect, macrophyte or algae, has been used intermittently with various systems of indicator species or community structure indices to rank lakes according to their trophic state.

Chlorophyll *a* concentration in lake water has been widely used in trophic classification systems because it is a relatively simple measurement of phytoplankton biomass. However, there are several weaknesses inherent with the use and interpretation of chlorophyll *a* data in classifying lakes including: (1) differing chlorophyll *a* cell volume ratios among algal genera and species and among the same species depending on environmental conditions and time of the year, (2) differing extraction efficiencies among the groups of algae, and (3) interferences from various decomposition byproducts including pheophytin. A further complication is that a given chlorophyll *a* value is only indicative of the amount of algal biomass present in the lake water at the time and place of sampling; it sometimes may not be representative of a lake's average summer concentration due to the fluctuations in phytoplankton biomass common to Wisconsin lakes. Also, in some lakes there is doubt about the rationale of evaluating a lake's trophic status on the basis of chlorophyll *a* when the lake may be dominated by macrophytes. Trophic classification of a macrophyte-dominated lake based on water clarity or chlorophyll *a* concentration alone generally results in underestimating

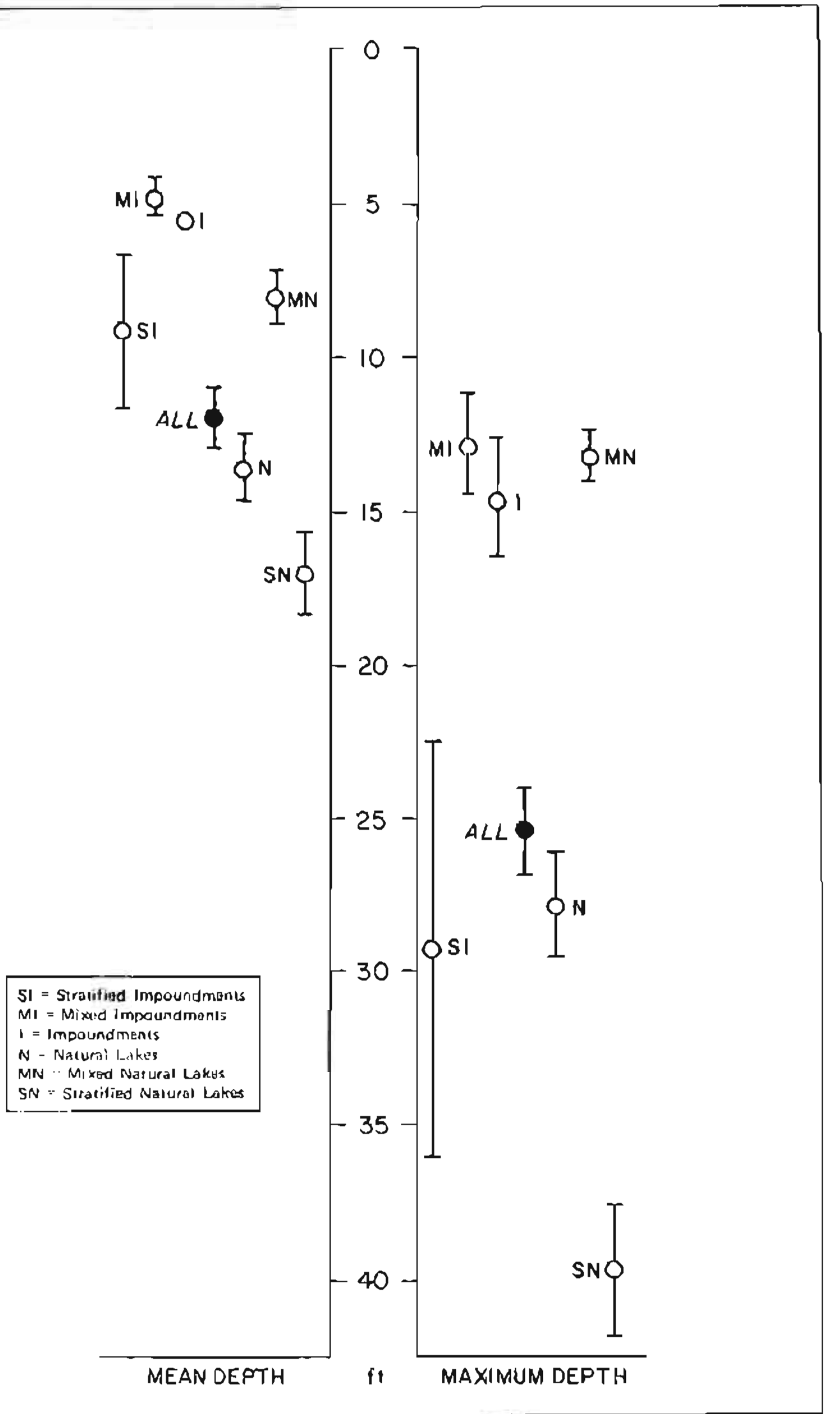


FIGURE 28. *Comparisons of mean and maximum depths (means ± 95% C.I.) for different types of lakes (random data set).*

the lake's productivity and trophic status. In some of these lakes, most available plant nutrients are believed to be used by macrophytes, thus limiting algal growth during the summer period. Another factor to consider is that the inhibit maximum algal production, thereby greatly affecting water clarity and chlorophyll *a* concentrations. Nevertheless, categorization of lakes based on chlorophyll *a* content is an accepted procedure, and lakes have been and no doubt will continue to be compared on the basis of seasonal or annual chlorophyll *a* concentrations.

The chlorophyll *a* data presented for Wisconsin lakes, based on the random data set, represent only one summer sampling per lake, which limits the interlake comparisons that can be made. However, the number and distribution of lakes within various concentration ranges, according to lake types, give a general representation of the trophic state of Wisconsin lakes. Selection of chlorophyll *a* concentration ranges appropriate to use in discussing water quality is based on comparisons of the relationship between chlorophyll *a* and water clarity for different lake types (Fig. 29). The correlation varies considerably between lake types and the establishment of a clearcut relationship between chlorophyll *a* concentrations and water clarity or perceived water quality, which would precisely describe all lakes, is not possible. However, field observations and comparisons of several subsets of our data provide a relationship of chlorophyll *a* concentrations to water clarity which can be used as a general index or guide to describe water quality of Wisconsin lakes (Table 19). These water quality index values are suggested for general comparative purposes and should not be construed as definitive standards, even though the values closely correspond to levels suggested by others (Vallentyne 1969, Michalski and Conroy 1972).

Mean chlorophyll *a* concentrations for different lake types were considerably higher than median values (Fig. 30); therefore, the percent distribution of lakes within various ranges better portrays the water quality of the state's lakes based on chlorophyll *a* (Fig. 31). As was the case with water clarity, the greatest number of lakes with poor water quality based on chlorophyll *a* concentrations were in the impoundment, drainage lake and mixed lake categories. Also, among stratified lakes, chlorophyll *a* level and water quality in the epilimnion were correlated with the D.O. level in the hypolimnion; 78% of the stratified lakes with chlorophyll *a* greater than 10 μg/l showed severe D.O. depletion in the hypolimnion.

Regional distributions of lakes classified according to chlorophyll *a* concentrations (Fig. 31) closely resemble the percentage distributions for water clarity (Table 20). The Southeast, Central and Northwest regions had slightly higher percentages of good and very good to excellent lakes when based on chlorophyll *a* instead of water clarity. This could be attributed to color interference in the Northwest Region lakes and possibly to the formation of calcium carbonate particles causing light reduction in the higher alkalinity Central and Southeast region lakes. Eighty-five percent of the lakes with very good to excellent water quality on the basis of chlorophyll *a* concentrations were located in the two northern regions, while 42% of the lakes with very poor water quality were found in the southern regions.

Potential Production

Another water quality parameter widely used in trophic classification is phosphorus. Total phosphorus is most often used, although some investiga-

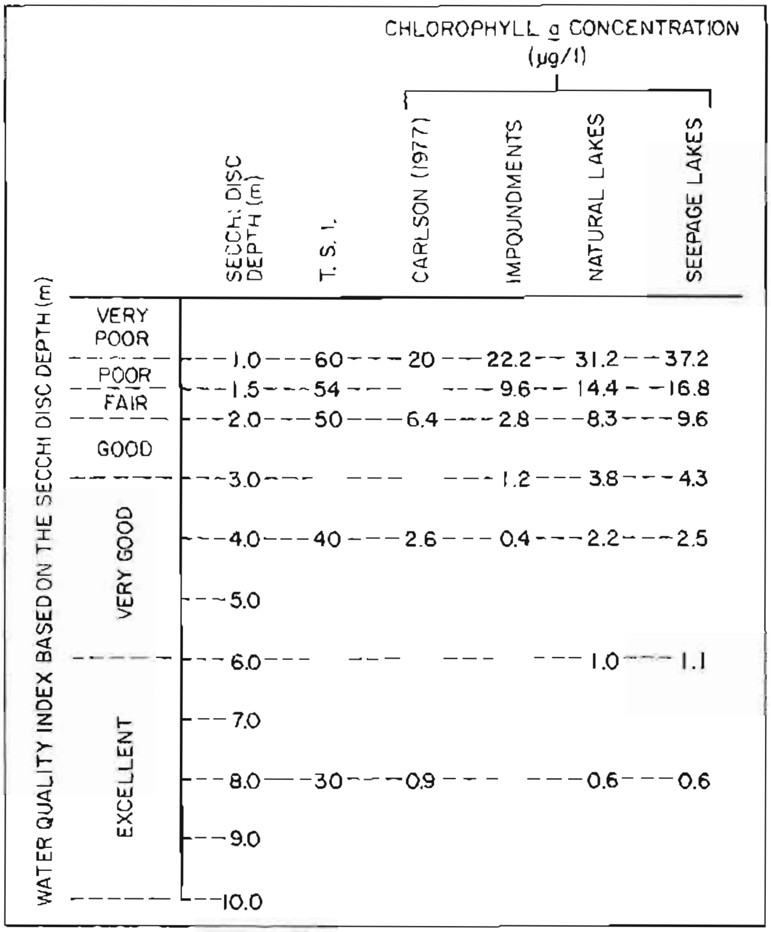


FIGURE 29. *Comparisons of chlorophyll* a, *water clarity, and Trophic State Index scales (Carlson 1977). (above)*

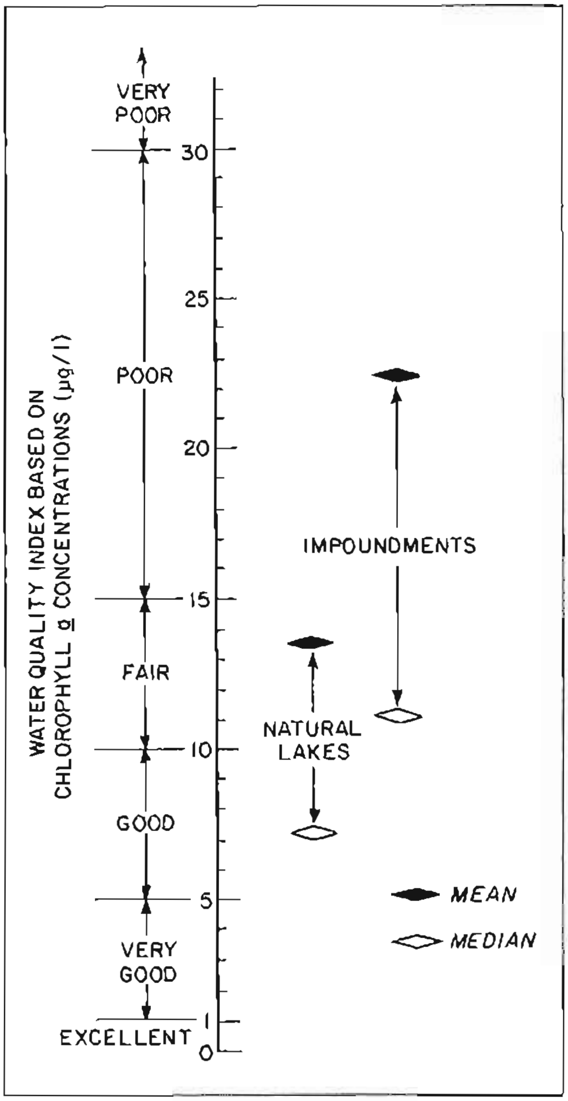


FIGURE 30. *Chlorophyll* a *concentrations for Wisconsin natural lakes and impoundments in relation to the water quality index. (right)*

TABLE 19. *Apparent water quality based on chlorophyll a and water clarity as related to the Carlson Trophic State Index.*

Chlorophyll *a* (µg/l)	Apparent Water Quality	Approximate Water Clarity Equivalent (m)	Approximate TSI* Equivalent
<1	Excellent	>6	<34
1-5	Very Good	3.0-6.0	34-44
5-10	Good	2.0-3.0	44-50
10-15	Fair	1.5-2.0	50-54
15-30	Poor	1.0-1.5	54-60
>30	Very Poor	<1.0	>60

*Based on Carlson (1977).

TABLE 20. *Comparison of percentages of lakes with different water quality based on water clarity and chlorophyll a concentrations for different Wisconsin lake regions (random data set).*

Region	Water Quality: Very poor	Poor and Fair	Good	Very Good and Excellent
Northeast	16-5*	26-22	22-38	36-34
Northwest	19-9	30-27	29-36	22-29
Central	5-0	29-21	39-44	26-35
Southeast	33-22	31-26	26-31	9-22
Southwest	61-40	32-47	7-13	0-0

* First number = percentage based on water clarity; second number = percentage based on chlorophyll *a* concentration.

tors have preferred reactive phosphorus or only biologically available phosphorus. Phosphorus is an essential element in the nutrition of aquatic plants (both macrophytes and algae) and therefore in-lake concentrations should, in theory, be related to the biological production and the trophic state of the lake. This interrelationship will be discussed in detail in the following section. Although there are a great many variables which may affect chlorophyll *a* concentration in individual lakes, there is generally a strong relationship between total phosphorus and chlorophyll *a* concentrations in Wis-

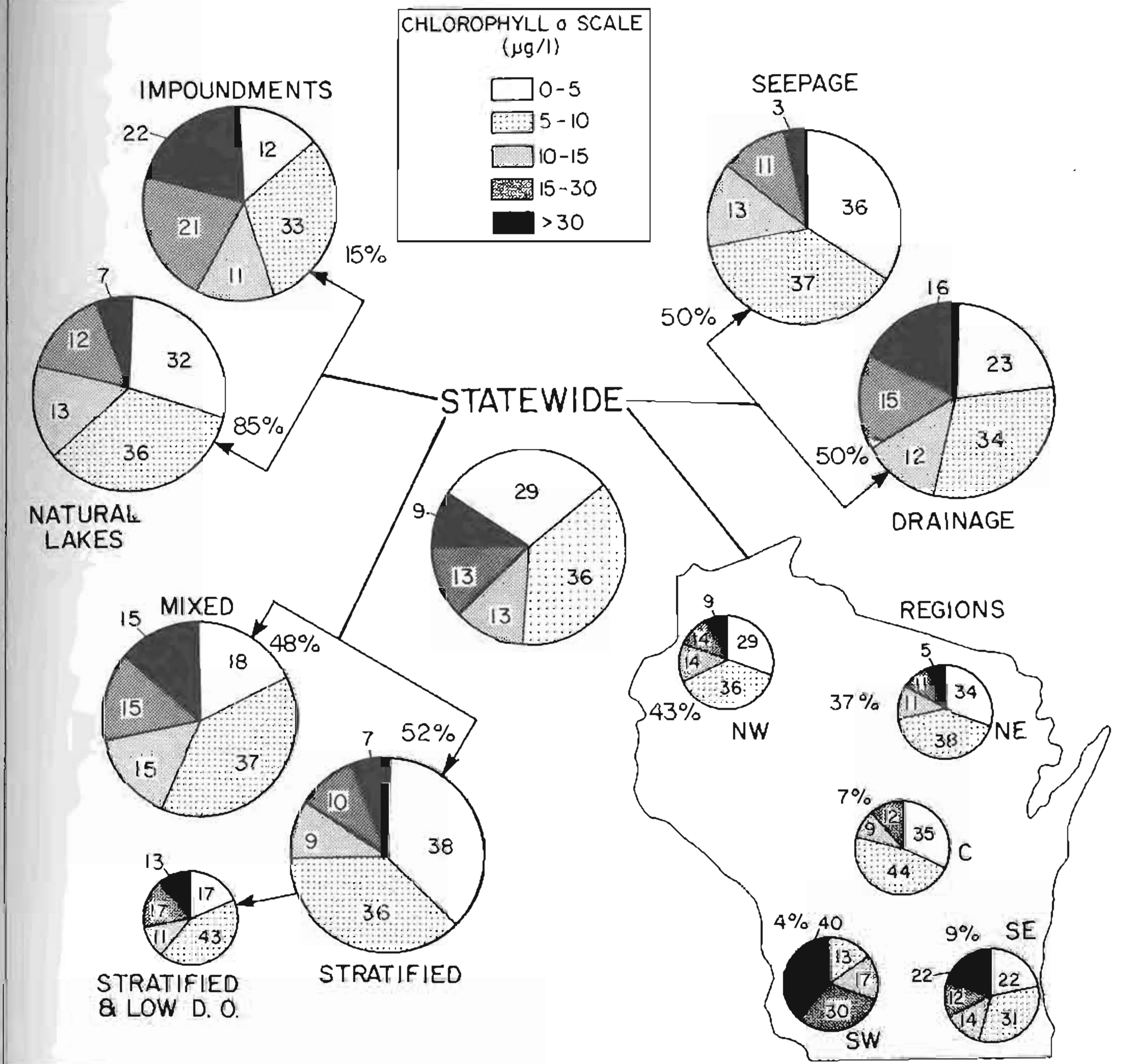


FIGURE 31. *Percent distribution of lakes within different ranges of chlorophyll a concentrations separated on the basis of lake type and region.*

consin lakes. On the basis of total phosphorus concentration, lakes may best be evaluated according to "potential" productivity, since phosphorus is not always channeled into biological production.

The estimated phosphorus levels required to produce various responses in chlorophyll *a* production and values given as separation criteria to describe different water quality conditions (Table 19) were based on the best-fit linear regression equation between chlorophyll *a* and total phosphorus (Fig. 32 and Table 26). The total phosphorus levels suggested as separation points for categorizing water quality of Wisconsin lakes are based on these chlorophyll *a* equivalents (Table 21). The total phosphorus-chlorophyll *a* relationship has to be used because total phosphorus concentrations are not "visible"; therefore, the relationship with chlorophyll *a* concentrations must be used as a cross-index to determine water quality. Again, it should be reiterated that many factors influence an individual lake's response to the amount of total phosphorus present (i.e., light and other nutrient limitation, toxic waste effects, flushing time or "washout", available forms of phosphorus, etc.) in addition to the factors affecting chlorophyll *a* fluctuations mentioned previously.

There was a good correlation between water quality indices based on total phosphorus (Fig. 33) and chlorophyll *a* (Fig. 30); the median for natural lakes and impoundments was nearly the same for both indices. Natural lakes were rated as having good overall water quality, while impoundments have a fair rating. However, while natural lakes maintain their overall good water quality rating based on water clarity, impoundments dropped to poor, probably due to the generally

TABLE 21. *Apparent water quality index for Wisconsin lakes based on water clarity, chlorophyll a content and total phosphorus concentrations (random data set).*

Water Quality Index	Approximate Water Clarity Equivalent (m)	Approximate Chlorophyll *a* Equivalent (μg/l)	Approximate Total Phosphorus Equivalent (μg/l)	Approximate TSI* Equivalent
Excellent	> 6.0	< 1	< 1	< 34
Very Good	3.0-6.0	1-5	1-10	34-44
Good	2.0-3.0	5-10	10-30	44-50
Fair	1.5-2.0	10-15	30-50	50-54
Poor	1.0-1.5	15-30	50-150	54-60
Very Poor	< 1.0	> 30	> 150	> 60

* After Carlson (1977).

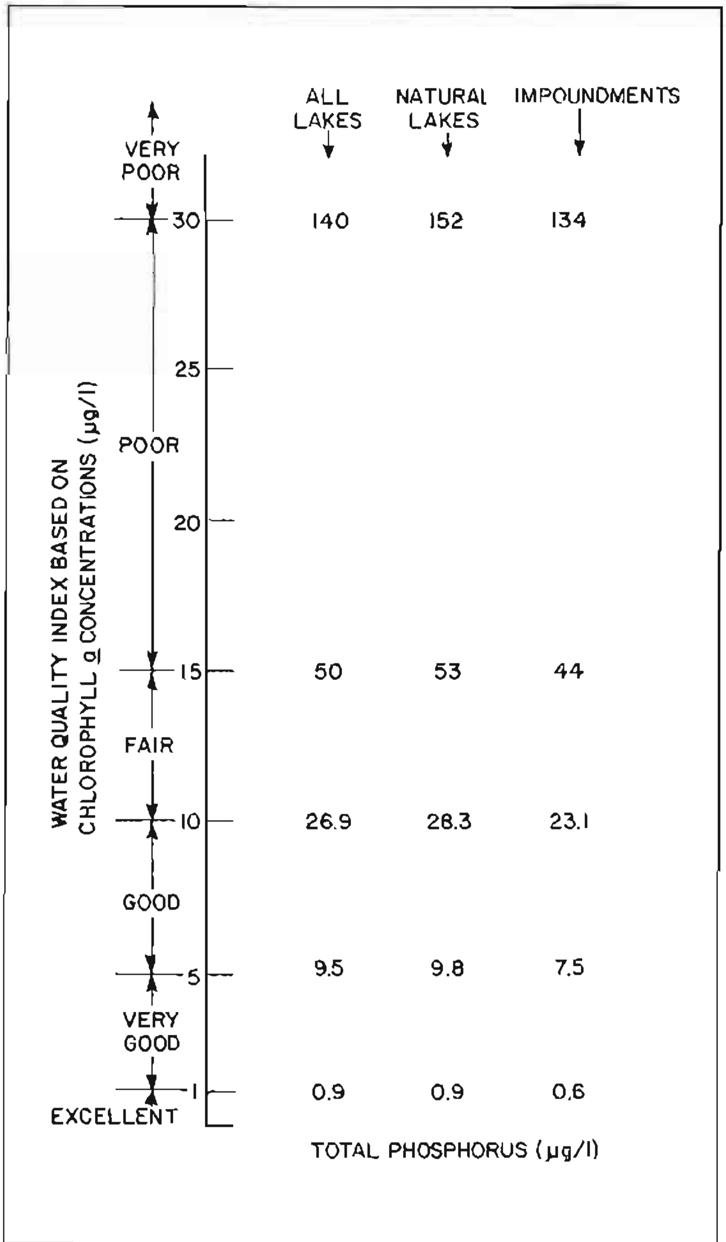


FIGURE 32. *Comparison of chlorophyll* a *and total phosphorus concentrations for different lake types (based on linear regression analysis) in relation to the water quality index.*

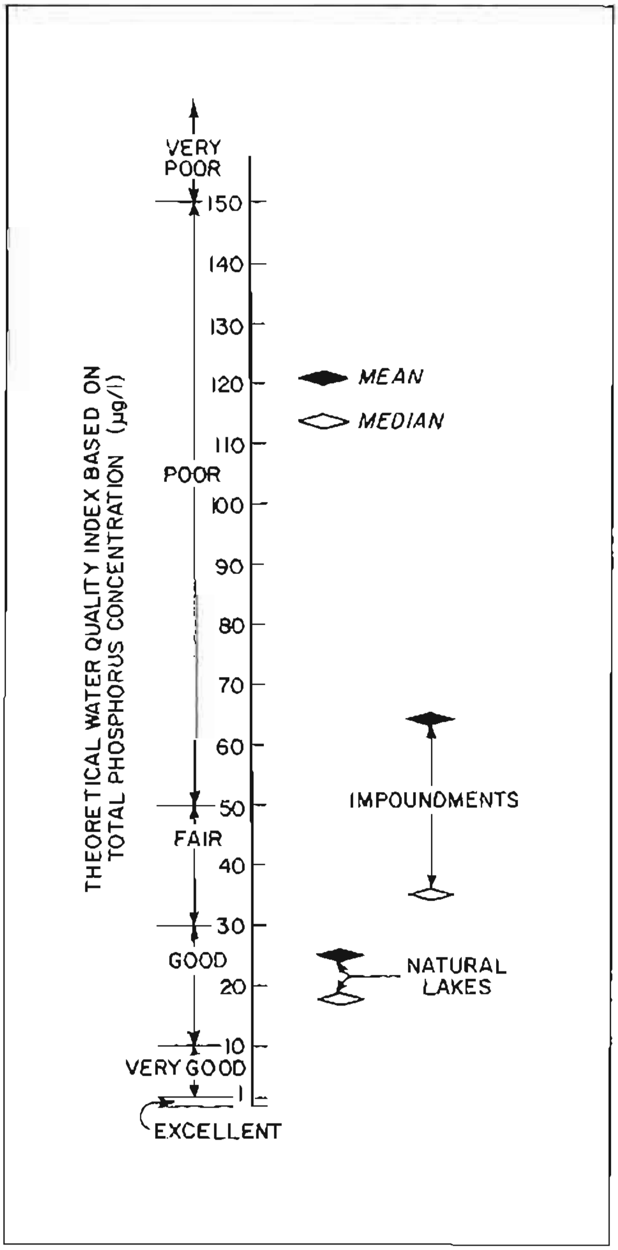


FIGURE 33. *Total phosphorus concentrations for Wisconsin natural lakes and impoundments in relation to "expected" water quality.*

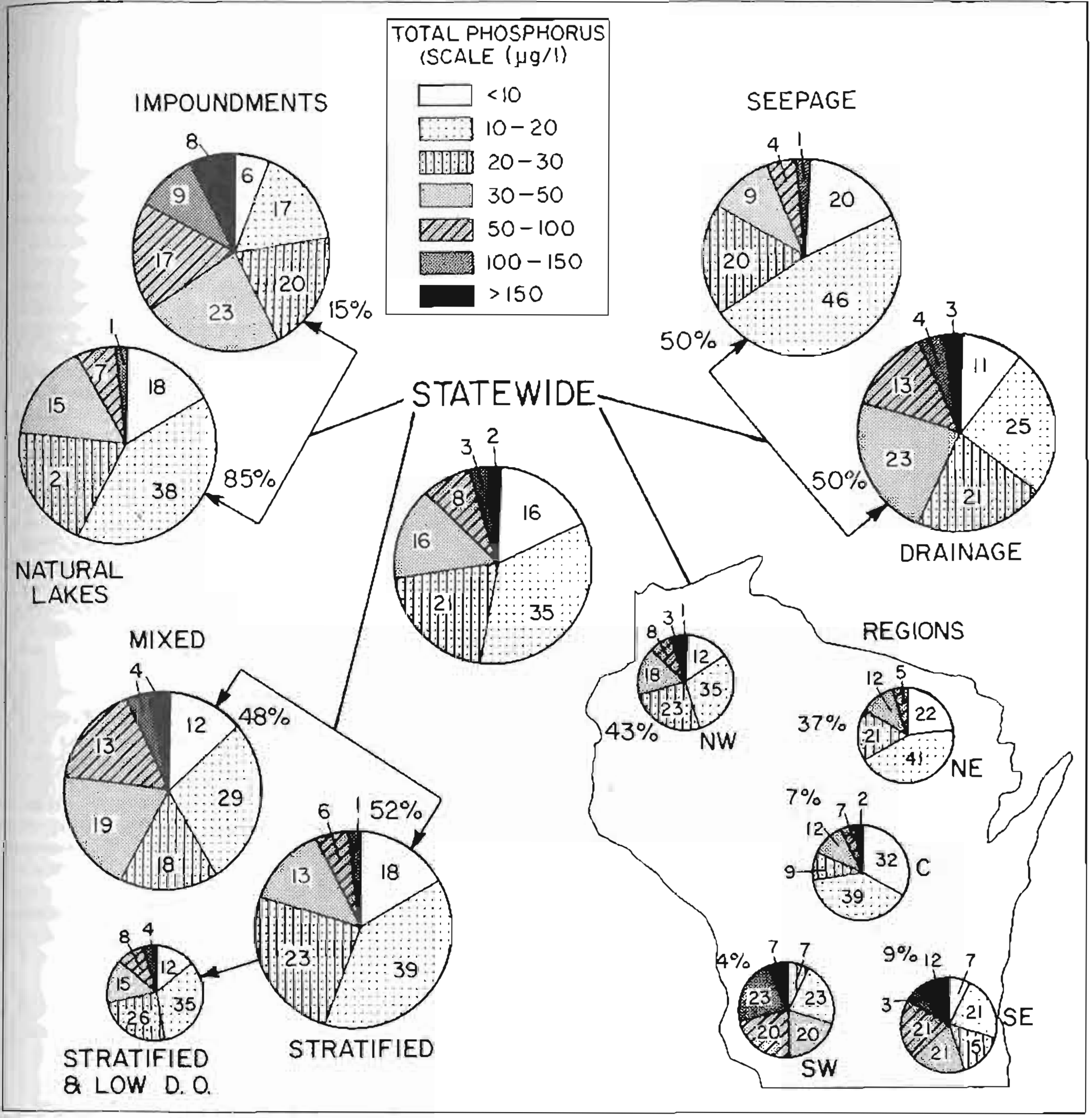


FIGURE 34. *Percent distribution by total phosphorus for different lake types and regions.*

higher nonalgal turbidities and color associated with drainage systems (Fig. 26). The mixed and drainage lakes and impoundments had higher percentages of poor and very poor water quality lakes than the stratified, seepage and natural lakes (Fig. 34).

Regional distributions were similar to those reported earlier based on chlorophyll *a* and water clarity. Of Southeast and Southwest lakes, 36% and 50%, respectively, were found to have poor water quality based on epilimnetic total phosphorus concentrations. This corresponds to 34% and 70% of the lakes in the same two regions where chlorophyll *a* concentrations indicated poor water quality (Fig. 31). A comparison of the percentages of lakes statewide classed as to water quality shows that slightly higher percentages of lakes may be classed as fair, poor, or very poor based on water clarity rather than on either chlorophyll *a* or total phosphorus (Fig. 35). Again, this is to be expected because of color and turbidity interferences. Limitations on chlorophyll *a* and total phosphorus measurements at the lower levels makes the water clarity scale the preferred index for clean, clear lakes, while the chlorophyll *a* index may be the most appropriate index at the other end of the spectrum (Sloey and Spangler 1978).

Comparisons between the rating of a lake on the basis of its chlorophyll *a* concentrations (actual production) and the total phosphorus concentration (potential production) could prove to be a valuable ratio in lake management.

The apparent water quality index provided in Table 21 should likewise be of great value to resource personnel involved in lake management. The descriptive narrative classes should make for easier, clearer communications between professionals and the interested public.

FACTORS AFFECTING LAKE TROPHIC STATUS

Water Clarity

Relationships

Water clarity-chlorophyll *a*. The water clarity-chlorophyll *a* association has been one of the most popular water quality associations used in the field of limnology recently. This relationship and the association with total phosphorus is of mutual interest to limnologists, fisheries biologists and water quality planners.

A review of the current literature concerning this topic produces an enormous number of papers which incorporate these parameters into various models to predict lake water quality (Deevey 1940, Dillon and Rigler 1975, Jones and Bachmann 1976, Williams et al. 1977, and Schnindler, Fee, and Ruszcynski 1978, to mention only a few). The development of these predictive equations is based on the assumption that water clarity is related to chlorophyll *a*, and that chlorophyll *a* is related to algal cell biomass.

Since Deevey's (1940) early work on Connecticut lakes, investigators have reported a wide range in relationships between chlorophyll *a*, total phosphorus and water clarity (see Davis et al. 1978, Dolan et al. 1978, and Nicholls and Dillon 1978 for further discussion concerning the reliability and variations inherent in these relationships). Some of these differences are due to factors other than chlorophyll *a* concentration. Numerous investigations have reported on the relationship be-

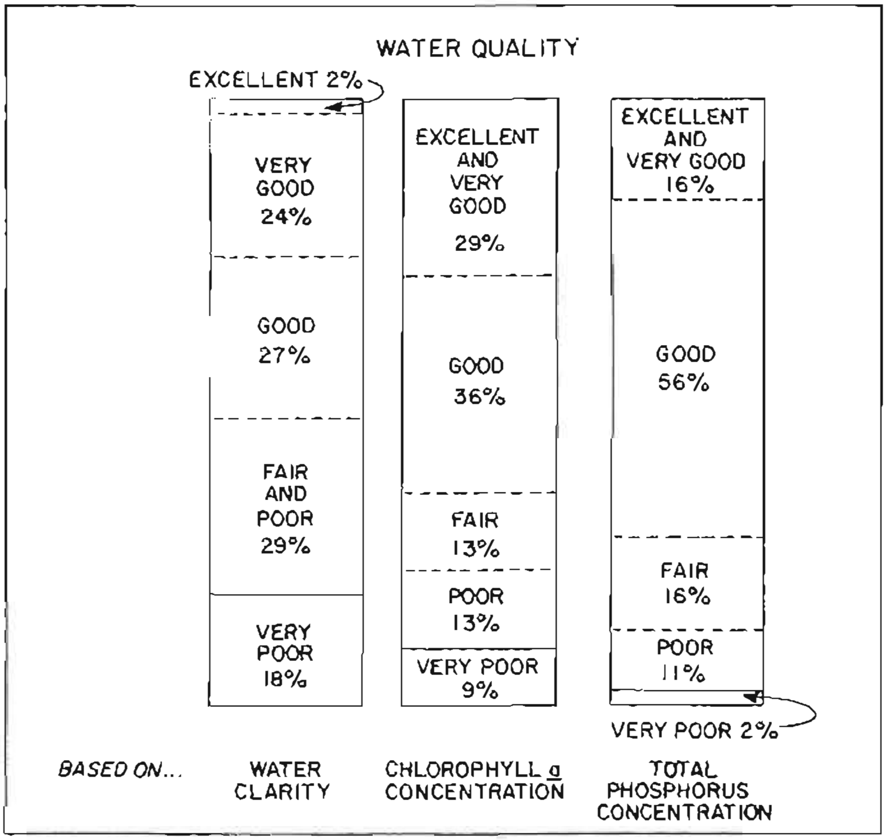


FIGURE 35. *Percentages of Wisconsin lakes in different water quality classes based on three water quality parameters. (top)*

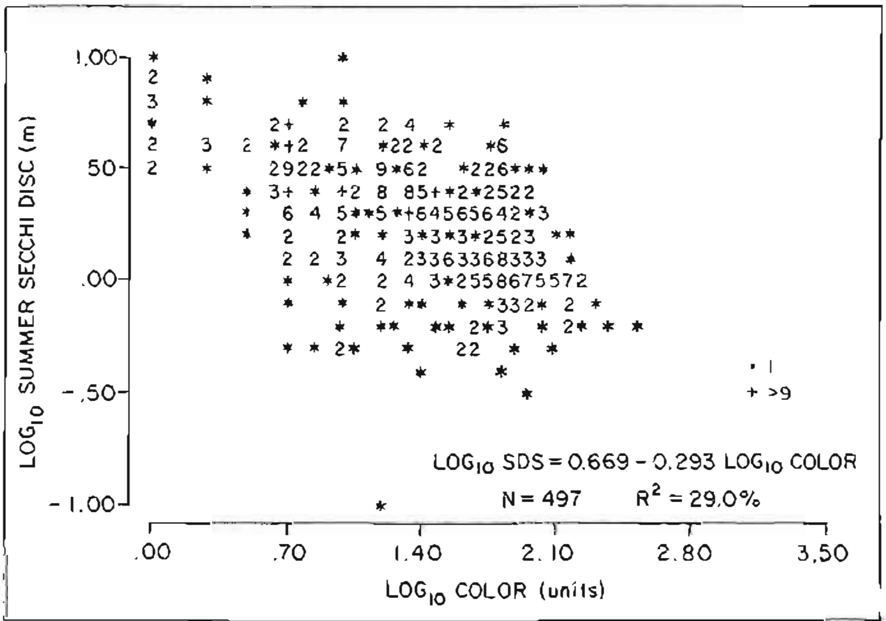


FIGURE 36. *Plot of Log-transformed water clarity and color data for 497 Wisconsin lakes (random data set). (bottom)*

TABLE 22. *Comparison of linear regression equations for water clarity (random data set).**

Wisconsin Lakes		Florida Lakes (Brezonik 1977)	
$Log_{10}SDS = 0.759 - 0.517\ log_{10}chla$ (µg/l)	$R^2 = 53.3\%$	$Log_{10}SD = 0.63 - 0.55\ log_{10}chla$	$R^2 = 75\%$
$Log_{10}SDS = 0.669 - 0.293\ log_{10}color$ (units)	$R^2 = 29.0\%$	$Log_{10}SD = 0.84 - 0.41\ log_{10}color$	$R^2 = 55\%$
$Log_{10}SDS = 0.466 - 0.547\ log_{10}turb.$ (JTUs)	$R^2 = 32.9\%$	$Log_{10}SD = 0.48 - 0.72\ log_{10}turb.$	$R^2 = 53\%$
1/SDS = 0.442 + 0.015 chla	$R^2 = 75.6\%$	1/SD = 0.48 + 0.032 chla	$R^2 = 59\%$
1/SDS = 0.487 + 0.045 color	$R^2 = 8.8\%$	1/SD = 0.76 + 0.0019 color	$R^2 = 10\%$
1/SDS = 0.532 + 0.048 turb.	$R^2 = 9.8\%$	1/SD = 0.44 + 0.12 turb.	$R^2 = 71\%$
1/SDS = 0.283 + 0.0163 chla + 0.0038 color	$R^2 = 74.5\%$	1/SD = 0.36 + 0.03 chla + 0.001 color	$R^2 = 63\%$
1/SDS = 0.363 + 0.0145 chla + 0.0280 turb.	$R^2 = 79.0\%$	--	
1/SDS = 0.339 + 0.0048 color + 0.0044 turb.	$R^2 = 20.6\%$	1/SD = 0.106 + 0.0025 color + 0.128 turb.	$R^2 = 89\%$
1/SDS = 0.209 + 0.0155 chla + 0.004 color + 0.0245 turb.	$R^2 = 78.1\%$		

*SDS = Secchi disc summer (m) SD = Secchi disc (m) tween water transparency, phytoplankton abundance and/or biovolume, color, and turbidity.

Juday and Birge (1933) reported that color was more important than the amount of plankton present in determining the transparency of 530 northeastern Wisconsin lakes. Deevey (1940) reached a similar conclusion in his early investigations of Connecticut lakes. More recently, investigators have demonstrated a correlation between water clarity and chlorophyll *a* for particular data sets. Logarithmic transformations of the two parameters (the nature of light attenuation follows a logarithmic pattern) linearizes the naturally hyperbolic curve of the relationship (Sakamoto 1966, Bachmann and Jones 1976). Lorenzen (1980) and Megard et al. (1980), among others, have pointed out that the attenuation of light is dependent upon both chlorophyll (Kc) and the natural background attenuation of the water (Kw). The proportion of Kw and Kc may vary seasonally as well as from lake to lake. The importance of Kc is high in lakes with large volumes of phytoplankton, but the resolution of the Secchi disc measurement is reduced. In low chlorophyll *a* lakes, the Kw becomes more important (Megard et al. 1980). Kwiatkowski and El-Shaarawi (1977) reported interference in the water clarity-chlorophyll relationship in Lake Ontario, possibly due to the formation of calcium carbonate precipitate particles which created a "milky" water appearance. Davis et al. (1978) reported that color was two to three times more important than phytoplankton in determining the transparency of Maine lakes.

In a survey of Florida lakes, Brezonik (1978) showed that the relative importance of color on the water clarity reading was somewhat dependent on the turbidity but at such high concentrations the care and precision with which the Secchi disc measurement is made becomes much more critical. Edmondson (1980) stressed that the Secchi disc measurement is dependent upon the number of particles scattering light in addition to other factors.

Several differences are of significance between Brezonik's data and the results of this study (Table 22). Brezonik found that the logarithmic transformation resulted in a greater reduction of the variance in water clarity than did the reciprocal transformation. Our data indicated the reverse to be true for the water clarity-chlorophyll relationship, while the color-water clarity and turbidity-water clarity relationships were improved. The reciprocal of the water clarity depth was highly dependent on chlorophyll *a* in both studies, but turbidity was also very important in the Florida study. In

TABLE 23. *Relationship of water clarity and chlorophyll a concentration in lakes with different perceived water color.*

Water Color	Regression Equation*	R^2
Green	$Log_{10}SDS = 0.786 - 0.537 \log_{10}chla$	72.4%
Brown	$Log_{10}SDS = 0.474 - 0.310 \log_{10}chla$	21.4%
Blue or clear	$Log_{10}SDS = 0.638 - 0.159 \log_{10}chla$	6.0%

*SDS = summer Secchi disc reading (m).
chla = chlorophyll *a* concentration (µg/l).

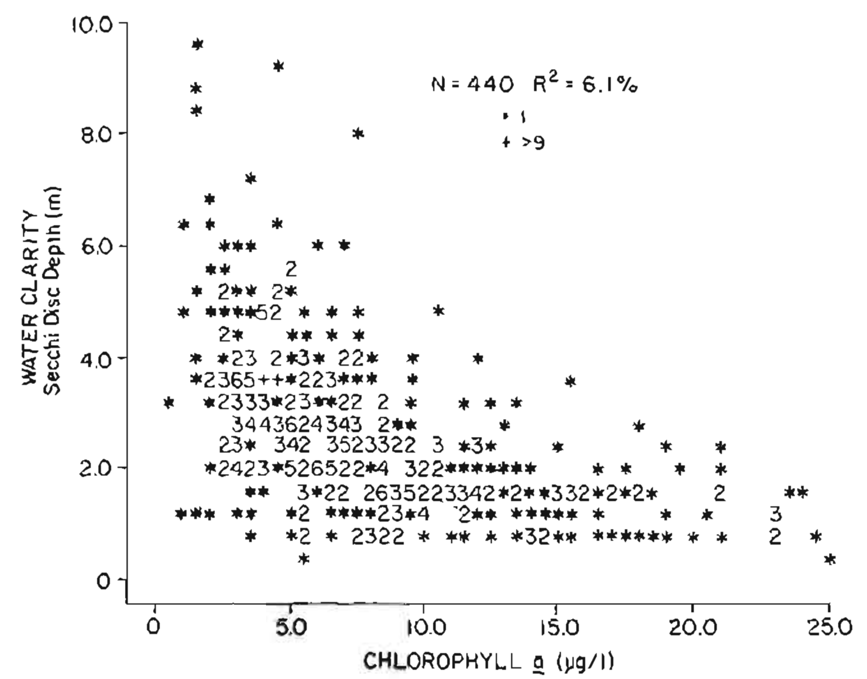


FIGURE 37. *Relationship of chlorophyll* a *concentration to water clarity in Wisconsin lakes with chlorophyll* a *less than 25 µg/l (random data set).*

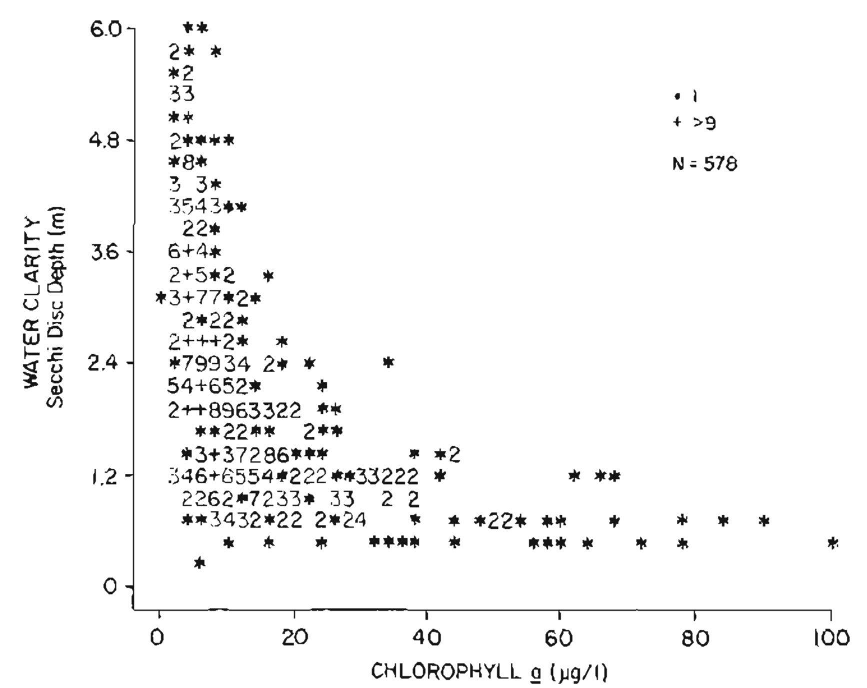


FIGURE 38. *Relationship between water clarity and chlorophyll* a *concentrations (random data set).*

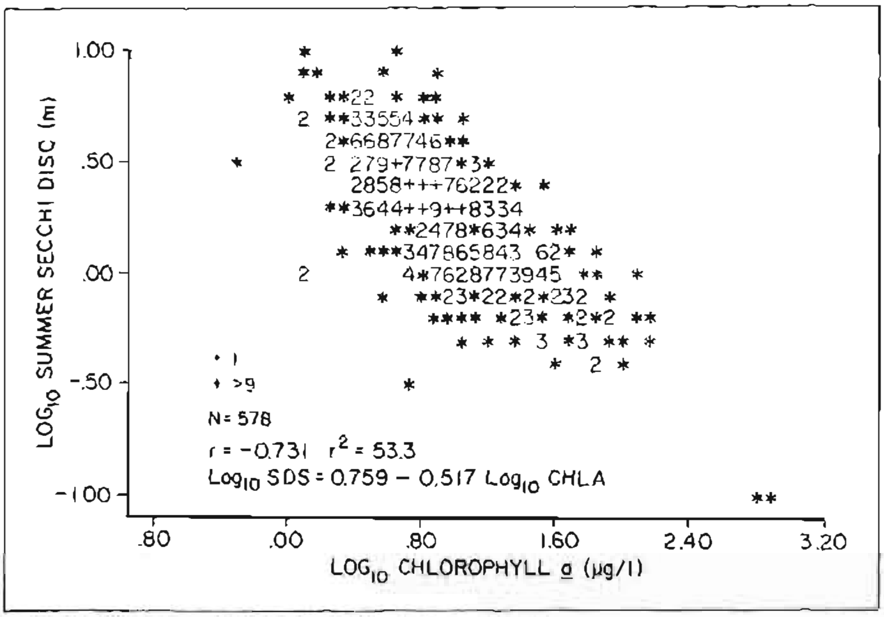


FIGURE 39. *Water clarity and chlorophyll* a *relationship in Wisconsin lakes (Log-Log transformation).*

fact, color and turbidity explained a higher percentage of the variance (89%) in the reciprocal of water clarity depth than did chlorophyll *a* and color (63%). The combined chlorophyll *a* and turbidity effect was not reported. Turbidity and color appear to be less important variables in influencing water clarity in Wisconsin lakes, as evidenced by low coefficients of determination (R^2) as determined by MINITAB analysis where R^2 equals the percent of the total variance explained by the regression corrected for N-2 degrees of freedom (Table 22). The relationship between water clarity and color has been previously discussed (see General Characteristics, pp. 29-30). For given individual color measurements, a considerable amount of variation can be expected (Fig. 36). Combining color or turbidity with chlorophyll *a* does little to improve the single chlorophyll *a*-water clarity relationship. A multiple regression of all three parameters only improves the explanation of variance by 2.5%. Part of the reason why there was only slight improvement may be the fact that turbidity and chlorophyll *a* are interrelated (see correlation matrixes in Appendix C). Even though the inverse water clarity transformation of Wisconsin lake data gives higher R^2 values, it appears that this is an artifact created by the distribution of a very few poor water clarity (high inverse Secchi disc) values in combination with the large number of low chlorophyll *a* values. Therefore, we believe this transformation is highly misleading and probably invalid. Reckhow (1979) and Shapiro (1979) present excellent examples and warnings concerning the use of such data transformations and presentations.

While measured color overall has been shown to be of minimal importance in affecting water clarity, at least in this study, apparent water color had a significant impact on the relationship between chlorophyll *a* and water clarity (Table 23). The R^2 value for green lakes is high while it is very low for clear or blue lakes and brown lakes. However, the latter two groups show higher R^2 values for color and water clarity.

The relationship between water clarity and chlorophyll *a* is described by a hyperbolic curve (Fig. 37, 38). A logarithmic transformation of both axes results in a linear relationship (Fig. 39) with a relatively small standard error (0.1928). This permits a fairly accurate estimation of *means* for water clarity or chlorophyll *a* for Wisconsin lakes within a particular range of values. Computation of the 95% confidence interval for the mean and individual values of water clarity demonstrates the practical application of the linear regression (Fig. 40). For example, all lakes with chlorophyll *a* levels equal to 10 µg/l ($Log_{10} = 1.0$) are predicted to have a mean water clarity reading ranging from 1.66 to 1.82 m. The wider band of lines represents the 95% confidence interval for water clarity for any single lake given a particular chlorophyll *a* reading. For this example, any *individual* lake with a chlorophyll *a* of 10 µg/l may be expected to have a water clarity anywhere from 0.69 to 4.12 m. The range in values of over 500% indicates that there is a lack of definition in the relationship and that the use of the regression equation to predict the water clarity of individual lakes is of extremely limited value. This variability in predictability decreases with increasing trophic status.

A comparison of our regression equations for the water clarity-chlorophyll *a* relationship (developed from 1976, 1977 and 1979 random survey data) with other published relationships (Fig. 40) shows that while all plots are somewhat similar, a considerable range of values may be predicted dependent upon which regression line is chosen. The Wisconsin data illustrate that considerable variation may occur within the same geographic region. This may be of considerable importance if attempts are made to equate trophic status of lakes based on one parameter compared to the other. For example, a 1.0-m Secchi disc reading may correspond to a mean chlorophyll *a* reading that ranges from 20-47 µg/l, depending on whether the Carlson (1977) equation or the 1976 Wisconsin lake data set is selected. It appears reasonable that in classifying lakes the equation which is believed to best fit the lakes being examined would be the one selected. However, many times it is difficult to select a refined equation on the basis of the information available, in which case literature values are chosen, resulting in some degree of error.

Regional differences in water clarity-chlorophyll *a* relationships occur not only on a large scale but also within rather restricted geographical regions. A comparison of regional subsets of the randomly sampled lakes shows that the Southwest Region differs considerably from other Wisconsin regions (Fig. 40). This is believed to be primarily due to the reduction in water clarity caused by high inorganic turbidities, and to high color in a few lakes in that region. By contrast, the Northeast Region lakes (which are generally relatively clear and less eutrophic) have an intercept of 1 µg/l which is very similar to that found by Carlson (1977).

The reasons for the differences found in the water clarity-chlorophyll *a* relationships for various data sets cannot always be definitely determined. Differences between data from different investigators (such as represented in Fig. 40) may be partially attributed to different methods of collection, analysis, and data compilation (Nicholls and Dillon 1978, Holm-Hansen 1978). However, even where differences due to these factors are negligible, such as is

FIGURE 40. *Linear regression lines for Log-transformed water clarity and chlorophyll* a *data for the randomly sampled lakes. (See Table 24 for number of lakes.)*

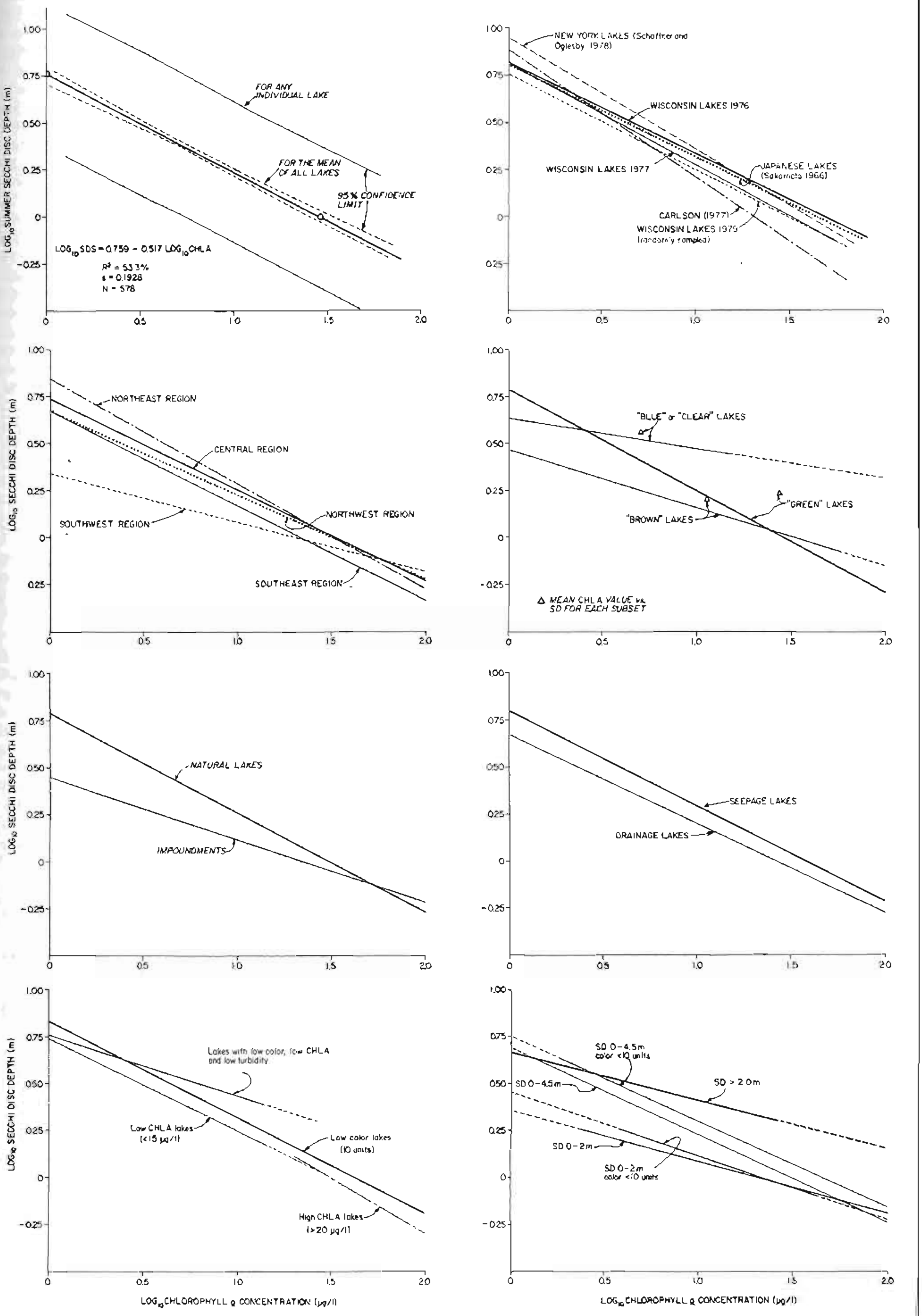

LOG10 SUMMER SECCHI DISC DEPTH (m)
FOR ANY INDIVIDUAL LAKE
FOR THE MEAN OF ALL LAKES
95% CONFIDENCE LIMIT
LOG10 SDS = 0.759 - 0.517 LOG10 CHLA
R² = 53.3%
s = 0.1928
N = 578
NEW YORK LAKES (Schaffner and Oglesby 1978)
WISCONSIN LAKES 1976
WISCONSIN LAKES 1977
JAPANESE LAKES (Sakamoto 1966)
CARLSON (1977)
WISCONSIN LAKES 1979 (randomly sampled)
LOG10 SECCHI DISC DEPTH (m)
NORTHEAST REGION
CENTRAL REGION
NORTHWEST REGION
SOUTHWEST REGION
SOUTHEAST REGION
"BLUE" or "CLEAR" LAKES
"GREEN" LAKES
"BROWN" LAKES
MEAN CHLA VALUE vs. SD FOR EACH SUBSET
NATURAL LAKES
IMPOUNDMENTS
SEEPAGE LAKES
DRAINAGE LAKES
Lakes with low color, low CHLA and low turbidity
Low CHLA lakes (<15 µg/l)
Low color lakes (10 units)
High CHLA lakes (>20 µg/l)
SD 0-4.5m color <10 units
SD 0-4.5m
SD > 2.0m
SD 0-2m
SD 0-2m color <10 units
LOG10 CHLOROPHYLL a CONCENTRATION (µg/l)

TABLE 24. *Linear regression equations expressing the relationship between Secchi disc and chlorophyll a for various restricted subsets of data.*

Subset Restrictions	Regression Equation*	N	s	R^2**
Seepage lakes[1]	Log_{10} SDS = 0.801-0.510 $\log_{10}$chla	275	0.1750	51.5%
Drainage lakes	Log_{10} SDS = 0.674-0.476 $\log_{10}$chla	301	0.1979	50.1%
Low color (< 10)	Log_{10} SDS = 0.838-0.515 $\log_{10}$chla	144	0.1729	53.6%
Valid SDS - chla (< 2 m)	Log_{10} SDS = 0.360-0.279 $\log_{10}$chla	264	0.1556	31.8%
SDS (< 2 m) and low color (< 10)	Log_{10} SDS = 0.457-0.346 $\log_{10}$chla	26	0.1383	44.9%
Epilimnetic SDS (0-4.5 m)	Log_{10} SDS = 0.753-0.459 $\log_{10}$chla	118	0.1520	53.2%
Epilimnetic SDS (0-4.5 m) and low color	Log_{10} SDS = 0.691-0.468 $\log_{10}$chla	535	0.1816	50.1%
SDS (2-9.5 m)	Log_{10} SDS = 0.671-0.257 $\log_{10}$chla	310	0.1271	20.7%
Low chla (< 15 µg/l)	Log_{10} SDS = 0.746-0.494 $\log_{10}$chla	365	0.2023	19.9%
High chla (21-800 µg/l)	Log_{10} SDS = 0.913-0.609 $\log_{10}$chla	91	0.1646	50.2%
Low color, low chla, low turb.	Log_{10} SDS = 0.769-0.327 $\log_{10}$chla	69	0.1528	17.0%
All natural lakes	Log_{10} SDS = 0.783-0.524 $\log_{10}$chla	489	0.1901	52.4%
All impoundments	Log_{10} SDS = 0.450-0.334 $\log_{10}$chla	87	0.1665	41.2%
Northeast Region	Log_{10} SDS = 0.844-0.557 $\log_{10}$chla	218	0.1792	50.9%
Northwest Region	Log_{10} SDS = 0.670-0.445 $\log_{10}$chla	241	0.2015	41.1%
Central Region	Log_{10} SDS = 0.734-0.480 $\log_{10}$chla	37	0.1662	34.6%
Southeast Region	Log_{10} SDS = 0.678-0.508 $\log_{10}$chla	54	0.1766	73.4%
Southwest Region	Log_{10} SDS = 0.337-0.259 $\log_{10}$chla	28	0.1617	20.6%

* SDS = Summer Secchi disc; chla = chlorophyll *a*.

** The low R^2 in some of the subsets may be partially the result of the elimination of data points above or below a given value along either the X or Y axis. Similarly, the slopes of the regressions may have also been affected.

[1] Elimination of one very high chla had no effect on regression.

s = standard error of estimate

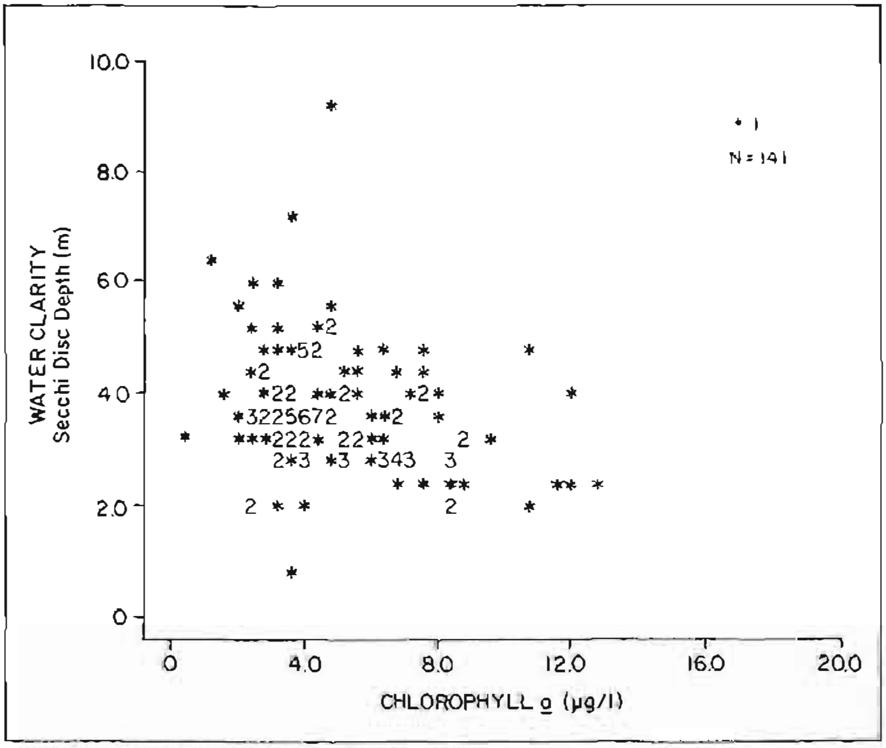


FIGURE 41. *Relationship between water clarity and chlorophyll* a *concentration in 141 "clear" lakes (random data set).*

the case in the Wisconsin regional comparisons, drastically different regressions can be obtained. The regressions and percentage variation that can be predicted through the use of the regressions vary considerably with the subset of lakes selected and their physical makeup or trophic status (Table 24).

The relationship of water clarity and chlorophyll *a* is dramatically different for lakes with different perceived water color (Fig. 40). Green lakes have the highest R^2 (72.4%) value and have the widest range of water clarity and chlorophyll *a* values. The R^2 values for brown and clear lakes are considerably lower, as are the slopes. Since 94% of the clear lakes had chlorophyll *a* concentrations less than 10 µg/l (see also Figs. 17, 41), nonalgal turbidity and color are relatively more important than chlorophyll *a* in affecting water clarity in the clear lake category. Brown lakes generally had poorer water clarity than green lakes with equal values of chlorophyll *a*. Figure 40 shows that color (or in some cases nonalgal turbidity) creates a significant decrease in water clarity readings where chlorophyll *a* concentrations are less than 30 µg/l. Caution is recommended in application of the data in Figure 40, since their use could lead to erroneous conclusions. For example, the actual mean chlorophyll *a* concentration of the green lakes (27 µg/l) is considerably higher than that which would be derived (10 µg/l) using the regression equation for green lakes with a mean Secchi disc of 1.75 m to predict mean chlorophyll *a* concentration. Part of this apparent discrepancy may be due to the log-log transformation and exclusion of Secchi disc values for a small number of shallow, high chlorophyll *a* lakes where the Secchi disc was visible on the bottom or disappeared in weed growth. These water clarity values were excluded from the data base, but their respective chlorophyll *a* values were included. Therefore, the mean values reported in Figure 40 represent the values for all green lakes with matching pairs of data points.

It appears that the relationship between chlorophyll *a* and water clarity in impoundments is seriously affected at lower concentrations of chlorophyll a by either color or nonalgal turbidity (Fig. 40). Because all impoundments are also drainage lakes, the two lake categories might be expected to be similar in respect to their relationships. Regressions of water clarity-chlorophyll a for seepage and drainage lakes are nearly parallel; water clarity is poorer in drainage lakes than seepage lakes with equal chlorophyll a concentrations (Fig. 40). This is in agreement with the findings reported earlier in General Characteristics (pp. 25-26), which show that chlorophyll a concentrations are not significantly different in seepage and drainage lakes, while water clarity, color and turbidity are different. These differences appear to be fairly uniform throughout the range of chlorophyll a values reported. The separation between the two lines may be attributed to generally higher color and turbidity (in this case specifically nonalgal turbidity).

The impact of color and turbidity on the water clarity-chlorophyll a relationship varies with the chlorophyll a concentration. While the overall water clarity-chlorophyll a relationship remains parallel for chlorophyll a values either above 20 µg/l or below 15 µg/l (Fig. 40), the R^2 value for low chlorophyll a lakes decreases by about 30% (Table 24). This would be expected due to the increasing importance of color and nonalgal turbidity at these levels, along with the increasing significance of accuracy in measuring chlorophyll a. Elimination of lakes with turbidities greater than 2 JTUs and color greater than 10 units from the low chlorophyll a subset did not improve the R^2 values (Table 24). However, the regression line did change, indicating that color and turbidity are important.

Lakes with similar low chlorophyll a levels but with no color or turbidity interference have generally better water clarity. Lakes with low color (< 10 units) have a regression line parallel but above that for either high or low chlorophyll a levels (Fig. 40). The R^2 value (53.6%) is high, indicating an improvement over the individual chlorophyll a subsets (Table 24).

Because chlorophyll a measurements used in this study were generally composites taken from 0-2 m, it is possible that in some instances the chlorophyll a and Secchi disc measurements do not directly correspond. A number of Wisconsin lakes develop metalimnetic algae blooms, in which case the 0-2 m layer of the epilimnion may be quite clear and low in chlorophyll a. The development of the algae bloom at or near the thermocline could in isolated cases cause abrupt reduction in water clarity which may not correspond at all to the concentration of chlorophyll a in the 0-2 m layer. Also, surface blooms of certain buoyant blue-green algae may produce high chlorophyll a levels in the 0-2 m layer, and yet water clarity may remain quite clear beneath the surface scum. While neither of these situations is believed to affect a great number of lakes in this data set, and thus have little significance in the water clarity-chlorophyll a relationship, they may quite possibly explain a number of the outliers which do not fit the general relationship.

In order to investigate possible biases, Secchi disc and chlorophyll a data were separated based on Secchi disc depth (Table 24 and Fig. 40). Secchi disc readings considered to be valid

Agricultural activities in lake watersheds are believed to have caused water quality changes in many Wisconsin lakes, but there is little historical data upon which to determine these changes.

Many Wisconsin lake shorelines have become highly developed, which may contribute to lake water quality problems.

which correspond to actual chlorophyll *a* measurements are represented by the less than 2.0-m lines. The group of lakes with 0-4.5 m Secchi disc readings represent lakes in which the water clarity may be less than expected based on the 0-2 m composite chlorophyll *a* reading, which would exclude lakes with metalimnetic blooms. The greater than 2-0-m water clarity group includes the same group of lakes with the exclusion of lakes with high epilimnetic blooms, but also includes clear lakes and those lakes with possible metalimnetic algae blooms. Comparisons of the values in Figure 40 indicate that some biases may exist, since the water clarity of lakes with identical chlorophyll *a* values (3-15 µg/l) may be considerably different (compare lines representing water clarity greater than 2.0 m with water clarity less than 2.0 m, Fig. 40).

Elimination of lakes in the 0-2 m water clarity data set with color greater than 10 units increases the slope and R^2 value of the regression. Adding lakes with water clarity from 2-4.5 m to the data set (roughly corresponding to the epilimnion) improves the R^2 even more. Elimination of lakes with color greater than 10 units does little to either the slope or R^2, but raises the intercept slightly. Eliminating lakes with water clarity less than 2.0 m and adding the lakes with water clarity greater than 4.5 m changes the water clarity-chlorophyll *a* relationship dramatically. The slope is very similar to those lakes with water clarity less than 2.0 m, but the intercept is higher and the R^2 is lower (20.7%). Because many of these lakes had low chlorophyll *a*, low color and low turbidity, the regression line is very similar to that shown for all lakes with low chlorophyll *a*, color and turbidity (Fig. 40).

The above analysis is important for two reasons; first, it further demonstrates the variability in the water clarity-chlorophyll *a* relationship, which is so widely used and acclaimed. Second, it illustrates the manner in which the relationship is affected or influenced by the particular types of data used in it. The relationship of phosphorus with water clarity through its influence on chlorophyll *a* production will be discussed in the next section.

Others. The impact of other factors on water clarity are less obvious, but correlations indicate that a number of other parameters are also associated with water clarity (Append. B). For instance, water clarity and pH were positively correlated in impoundments and negatively correlated in lakes with color levels less than 40 units. Kwiatkowski and Roff (1976) reported a strong negative correlation between water clarity and pH in northern Ontario lakes with pHs below 6.0. The significance of color is further indicated

TABLE 25. *Seasonal changes in Secchi disc (in meters) in different lake types.*

	Spring			Summer			Fall		
Lake Type	No.	Mean	SD*	No.	Mean	SD	No.	Mean	SD
Impoundments									
Mixed	67	1.19	0.54	55	1.11	0.66	68	1.32	0.64
Stratified	19	1.42	0.70	15	1.55	0.87	19	1.83	0.99
Natural Lakes									
Seepage									
Mixed	37	2.43	1.14	34	2.40	1.20	32	2.78	1.33
Stratified	106	3.26	1.56	82	3.13	1.68	112	3.38	1.61
Drainage									
Mixed	51	1.83	1.03	33	1.47	0.97	48	1.79	1.01
Stratified	120	2.82	1.29	79	2.61	1.49	123	2.93	1.42

*Standard deviation.

by the fact that water clarity was strongly correlated with all other parameters in low color lakes.

On a regional basis, fairly consistent associations of water clarity with turbidity, nitrogen and phosphorus were found. Water clarity was negatively correlated with calcium in the Southeast and Central regions and with alkalinity in the Southeast Region. This is consistent with other studies which have demonstrated that excess calcium in the form of colloidal particles may affect the penetration of light in the water column (Hutchinson 1975, Kwiatkowski and El-Shaarawi 1977).

Seasonal changes

Seasonal changes in water clarity are related to a number of factors. Water clarity observations show that Wisconsin lakes are generally clearest in winter, although algae blooms do sometimes occur under the ice. Lake type and stratification condition are important in describing seasonal changes in water clarity. Stratified lakes generally show greater fluctuations in water clarity from spring to summer than mixed lakes (Table 25), which is the inverse of changes occurring in total phosphorus concentration. At the higher trophic levels, lakes with poor water clarity may exhibit wide differences in total phosphorus content, while at the other end of the spectrum small differences in total phosphorus may be accompanied by drastic differences in water clarity.

Seasonal changes in water clarity are affected by the relationship of water clarity to nutrient dynamics and the overall trophic condition of lakes. Because mixed lakes generally are shallower and more eutrophic than stratified lakes, incoming nutrients accumulate and remain available to the biological system throughout the summer, while in many stratified lakes some of the nutrients present during the spring turnover and those nutrients entering the lake thereafter settle to the hypolimnion where they are generally unavailable until fall turnover. (Some transport of nutrients from the hypolimnion to the epilimnion occurs as the thermocline migrates downward in late summer.) Loading rates, lake volume, percentage bottom area exposed to epilimnetic mixing and resuspension, weather conditions, and biological activity all influence the summer phosphorus dynamics of a stratified lake and in turn greatly influence water clarity (Edmonson 1972, Barica 1974, Stauffer 1974, Fee 1979).

Water clarity generally varies more in stratified lakes with lower overall phosphorus levels than in mixed lakes where nutrient levels are higher. In mixed lakes, nutrient levels are usually high enough to maintain high levels of algal production, or nonalgal turbidity levels are sufficiently high during the summer and other seasons to maintain poor water clarity conditions throughout the year.

As would be expected, seasonal water clarity data (means) for different lake types indicate that water clarity generally is poorest during the summer season, deteriorating from spring to summer and then improving again from summer to fall (Table 25). While this is generally true, the number of lakes used in computing the summer means is less than either the spring or fall sample sizes due to the elimination of lakes whose water clarity improved to the point that the lake bottom was visible during the summer or growths of macrophytes prevented accurate water clarity measurements. When only lakes with corresponding sets of spring and summer water clarity data

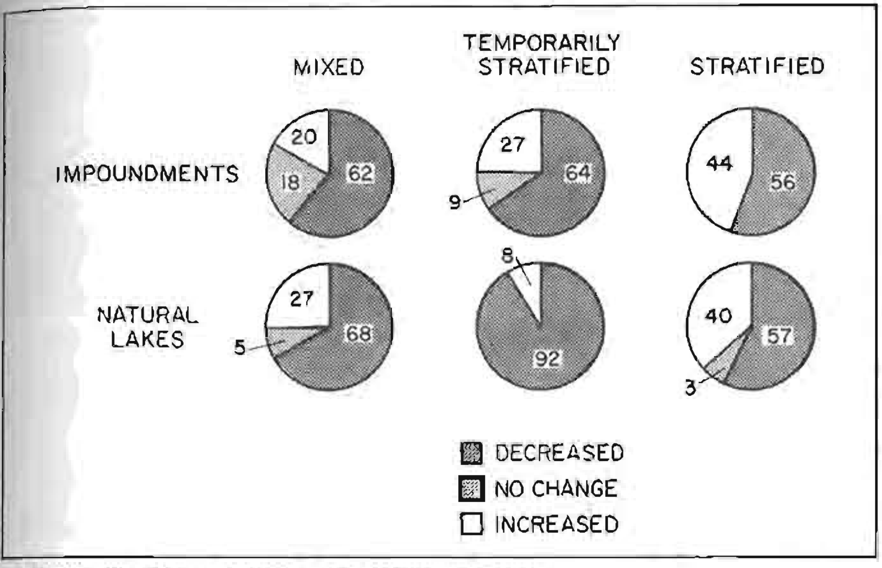


FIGURE 42. *Changes in water clarity from spring to summer as related to lake type.*

were analyzed, a slightly higher percentage of stratified lakes show improvement in water clarity from spring to summer (Fig. 42).

Summary

Generally, color and nonalgal turbidity appear to be important, but not as important as chlorophyll *a* in determining water clarity in Wisconsin lakes. In lakes with water clarity less than 2.0 m and chlorophyll *a* less than about 30 µg/l, color and/or nonalgal turbidity rather than chlorophyll *a* concentration seem to be the most important causes for the reduced water clarity. Lakes with chlorophyll *a* less than 10 µg/l, accompanied by low color and low turbidity, or lakes with water clarity greater than 2.0 m, are likely to show water clarity-chlorophyll *a* relationships which are considerably different than those for more eutrophic or colored lakes. Our data show that the predictive capabilities of the water clarity-chlorophyll *a* relationships are generally unreliable for Wisconsin lakes on an individual lake basis, but the relationships are suitable for making generalizations about different lake types and characteristics.

Seasonal changes are most pronounced in stratified lakes of low trophic state (good water quality), since relatively small changes in nutrients can greatly influence chlorophyll *a* concentrations at low levels. Conversely, slight changes in nutrient levels in more eutrophic lakes may produce no apparent change in water clarity; in these lakes, great changes in nutrient concentrations may be required to produce a noticeable change in perceived water clarity or quality.

Chlorophyll *a* Concentration (Algae Production)

The assessment of lake trophic status through the use of chlorophyll *a* concentrations is a common practice, but it is not necessarily a completely valid one. In this study, chlorophyll *a* concentrations represent the condition of the open water (pelagic zone) at 0-2 m depth at the time of sampling. In some cases, the pelagic zone chlorophyll *a* may not accurately represent the lake's true trophic state due to the dynamics of the phytoplankton or macrophyte community. Vertical and horizontal differences in the distribution of phytoplankton populations may cause error in the estimation of a lake's trophic state. Metalimnetic or floating bluegreen algae blooms or windblown accumulations of algae may at times affect the samples collected and hence affect the overall perception of a lake's condition. In addition, previous investigators have found that chlorophyll *a* content per phytoplankton cell (on a per unit biovolume basis) may vary from season to season (Nicholls and Dillon 1978; see also Carlson 1980 for further discussion). In lakes with dense macrophyte growth, chlorophyll *a* concentration is sometimes relatively low, which can lead to underestimation of trophic status based on chlorophyll *a*. These and other problems recognized in interpretation of chlorophyll *a* data will be discussed further.

Chlorophyll *a* — Total Phosphorus

The water clarity-chlorophyll *a* relationship and chlorophyll *a*-total phosphorus relationship have been used to rank lakes according to their trophic status (Carlson 1977, Sloey and Spangler 1978) and in the eutrophication modeling process (Kirchner and Dillon 1975, Chapra and Tarapchak 1976, Larsen and Mercier 1976, Hutchins 1977, Ostrofsky 1978, Reckhow 1978*b*, Tapp 1978, Reckhow 1979, Smith 1979, and Ciecka, Fabian, and Merilatt 1980, to mention only a few). Even though Carlson stressed that the trophic state index number for a given lake could vary considerably depending on the time of the year sampled, the parameter chosen, and various interferences, many investigators have nonetheless used the Trophic State Index system in lake classification. Generally, these relationships have some legitimate value as long as the broad dispersion displayed by the data is kept in mind.

As Shapiro (1978) aptly stated, "Too great reliance on these relationships may give us a false sense of confidence and make us believe we know more than we do."

However, if the relationships are kept in their proper perspective, some important information can be gained from them.

The many factors affecting the water clarity-chlorophyll *a* relationship were previously discussed; other factors which influence the interrelationship of chlorophyll *a* with nutrient concentrations are shown in Figure 43. The major assumptions which tie the relationships together are that the amount of chlorophyll *a* present is primarily related to the phosphorus concentration and that the water clarity is in turn primarily dependent on the chlorophyll *a* concentration.

Although this is not always the case (see Lorenzen 1980, Megard et al. 1980), a plot of our water clarity-chlorophyll *a*-total phosphorus relationship for 1976-77 for Wisconsin lakes is evidence that the three parameters are generally related (Fig. 44). The relationship shows a considerable amount of scatter, even though some of the scatter may be explainable. A large number of the lakes with high total phosphorus levels found in the upper corner of Figure 44 had low N:P ratios (<15), and some of the other lakes with relatively high total phosphorus values were known to have received treatments of sodium arsenite for aquatic plant control. Arsenic interference in the total phosphorus analysis in some of these lakes may have caused erroneously high total phosphorus values (Lueschow 1972, Office of Inland Lake Renewal 1978).

Many of the lakes falling below and to the left of the general distribution pattern had either high color or high inorganic turbidities, while many of the

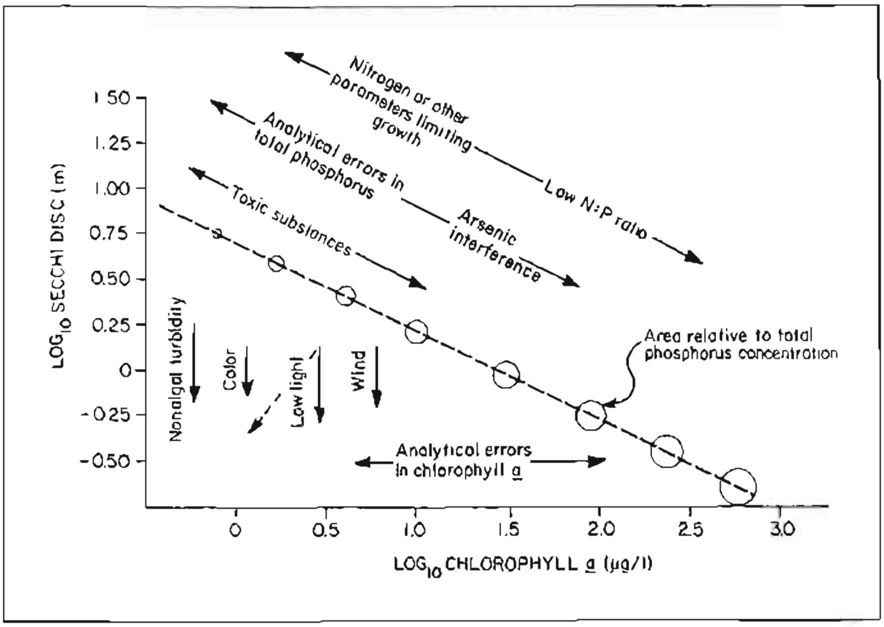


FIGURE 43. *Generalized diagram of water clarity-chlorophyll* a *-total phosphorus relationship demonstrating the effect of various interfering factors.*

lakes above or to the right of the general pattern were among the clearest lakes in the state. Some of the other incongruities may be related to weather conditions at or shortly preceding the time of collection. When all of the factors affecting the precision of the chlorophyll *a*, total phosphorus and water clarity measurements are combined, it is not too surprising to find a great deal of scatter in the relationship of these three parameters.

In the random and total data sets, the total phosphorus-chlorophyll *a* relationship also shows considerable variation (Fig. 45). The logarithmic transformation of these data also shows the variation (Fig. 46); the transformation actually results in a slight reduction in the explained variability over the untransformed data (-1.5%).

The relative ineffectiveness of the relationship as a predictive tool for individual lakes is illustrated by the broad bands of the confidence intervals about the regression line (Fig. 47).

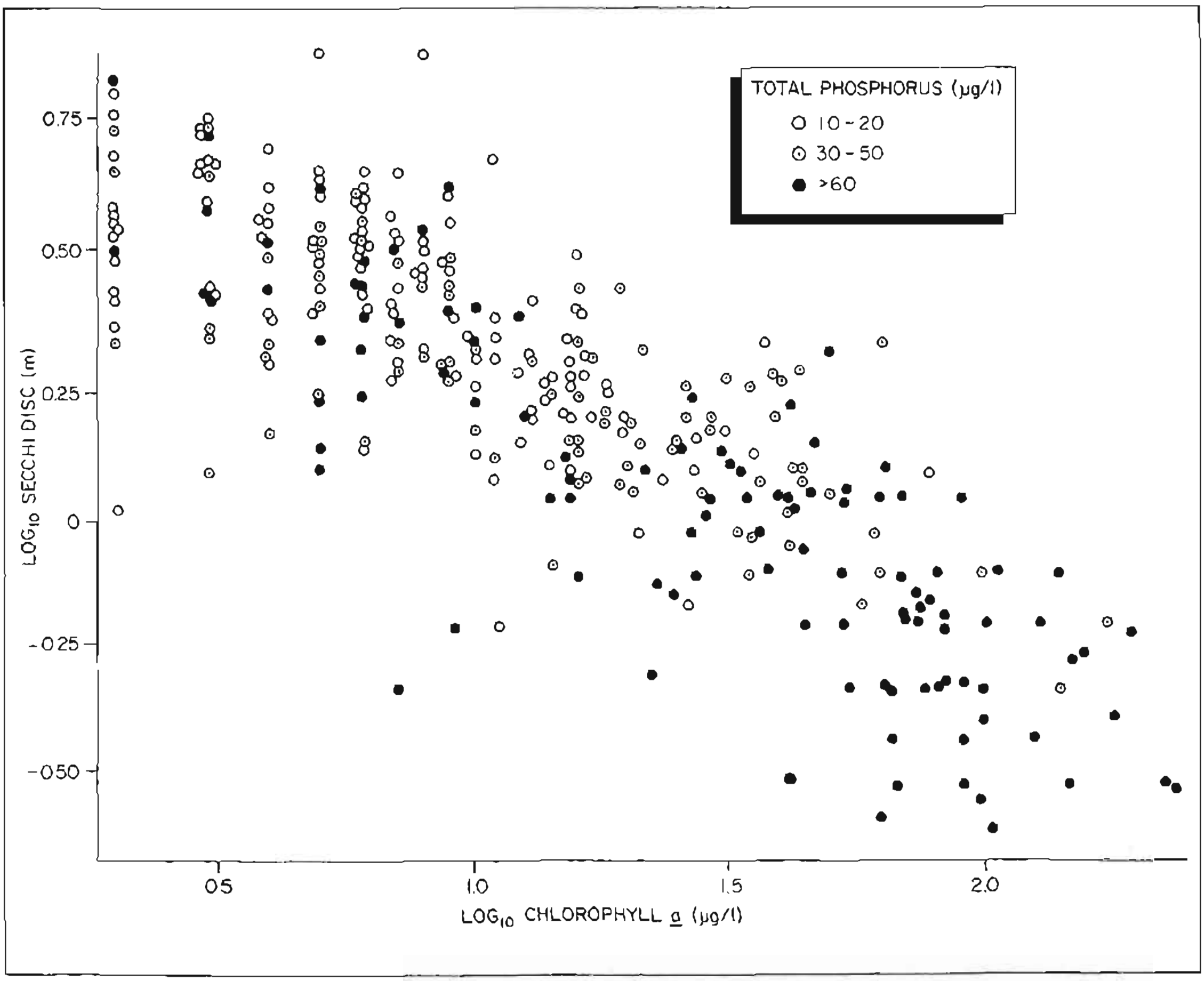


FIGURE 44. *Total phosphorus concentrations of lakes in relationship to their water clarity and chlorophyll* a *concentrations (based on 1976-77 Wisconsin lake data).*

Many other investigators have reported very good correlations between total phosphorus and chlorophyll *a* (Deevey 1940, Sakamoto 1966, Dillon and Rigler 1974, Jones and Bachmann 1975 and 1976, Lambou et al. 1976, Williams et al. 1977, Schindler 1978, Bartsch and Gakstatter 1978, and Oglesby and Schaffner 1978, to mention only a few). Nicholls and Dillon (1978) provided excellent comparative descriptions of published chlorophyll *a*-phosphorus relationships and discussed factors that influence the relationships. Comparison of the regression equations describing the chlorophyll *a*-total phosphorus relationship as derived from several different studies demonstrates the difficulties encountered in interpreting the effects of different variables on the relationship.

Regressions describing the relationship of chlorophyll *a* to total phosphorus for two sets of Wisconsin lake data (Fig. 48) are similar to the regressions reported for 418 northeastern U.S. lakes sampled by the U.S. Environmental Protection Agency (Williams et al. 1977). High inorganic turbidity (and consequent light limitation) was thought to be a primary factor involved in the differences between the EPA regressions and the regressions of Dillon and Rigler (1974), Bachmann and Jones (1976), and Carlson (1977) (Lambou 1978 pers. comm.), but many other factors may also account for the apparent shortfall in chlorophyll *a* production per unit of total phosphorus. Hern et al. (1981) suggested that more importance should be given to understanding the factors influencing differences in the response rate of phytoplankton. They found that limitation of light penetration, nontotal phosphorus nutrient limitation, the ratio of biologically available phosphorus to available nitrogen, macrophyte competition, toxic substances, and hydrologic retention times were important factors influencing the amount of chlorophyll *a* present. In order to evaluate which factors might be important in controlling chlorophyll *a* concentrations in Wisconsin lakes, the available data sets were analyzed on the basis of various physical and chemical characteristics (Table 26).

The slopes of the chlorophyll *a*-total phosphorus regression lines for the five Wisconsin lake regions are quite different than other cited regressions (Fig. 49). Coefficient of determination (R^2) values are highest in the Southeast Region and poorest for the Northeast and Southwest regions. A comparison of regression line slopes seems to indicate that the Southeast Region lakes (with the highest slope value of 0.915) have the best response rate for the amount of total phosphorus present. However, the Southeast Region lakes generally have lower chlorophyll *a* levels than expected for the amount of phosphorus present (sodium arsenite interference?). Therefore, the slope of the regression line may be misleading. Further evidence of this is demonstrated by the low slope (0.395) and high chlorophyll *a*-total phosphorus ratios of the Central Region lakes. Thus, the slope of the regression line appears to be related to the *relative* rate of change in the chlorophyll *a* response within a particular lake subset and is not necessarily related to the actual or *absolute* chlorophyll *a* response. The Southwest Region has a large number of impoundments with high total phosphorus levels, high turbidities, and short retention times, which probably accounts for the low R^2 value and low slope (relative response rate) of the chlorophyll *a*-total phosphorus relationship. The Northeast Region has a large number of lakes with low total phosphorus which appears to significantly affect the R^2 and slope (Fig. 50 and Table 26).

The total phosphorus-chlorophyll *a* correlation appears to be a little better in impoundments than in natural lakes (Table 26); the absolute response as indicated by the amount of chlorophyll *a* produced at given total phosphorus

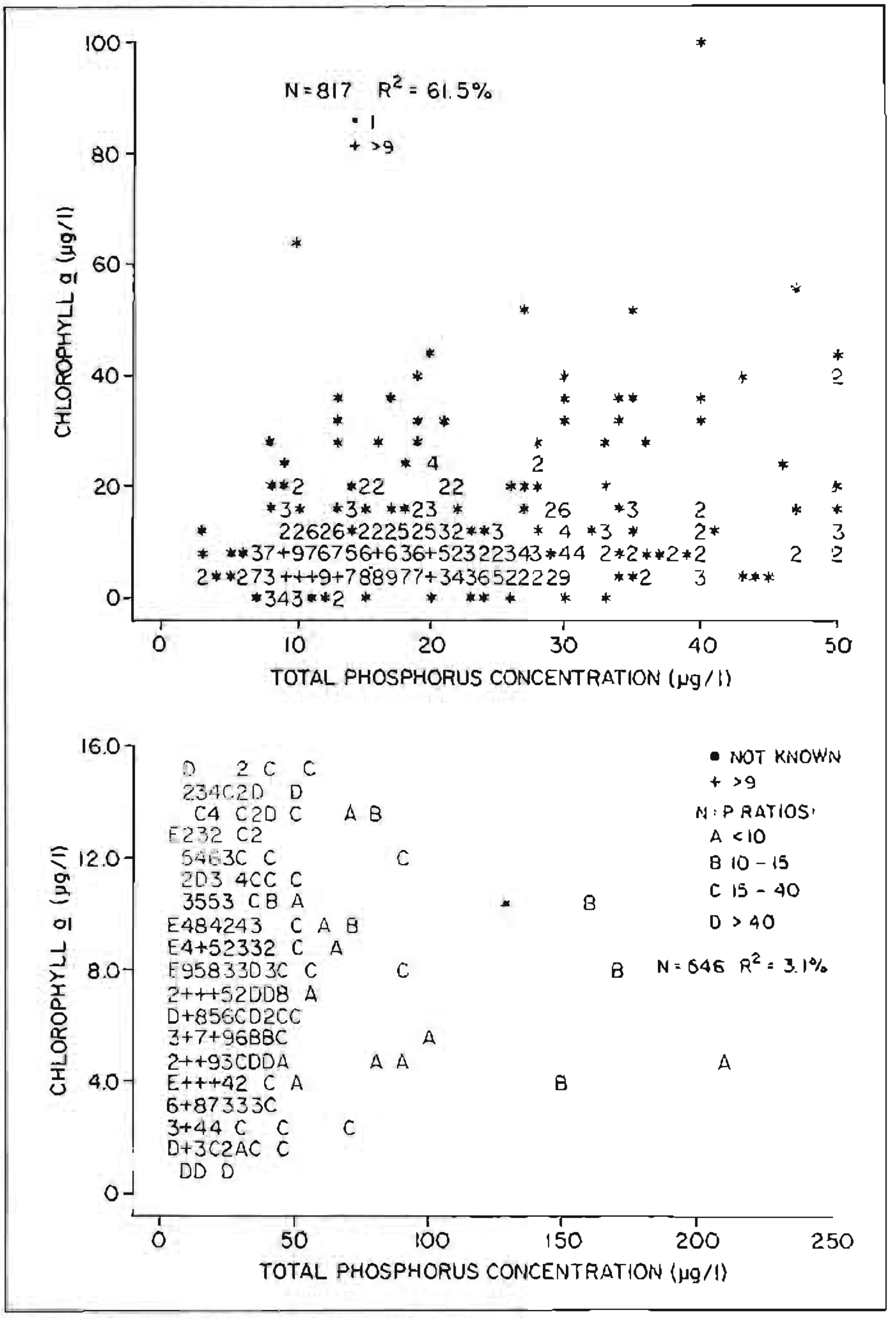


FIGURE 45. *Relationship of chlorophyll* a *and total phosphorus in Wisconsin natural lakes. (Top: total data set; bottom: random data set with N:P ratios given.)*

levels seems to be slightly higher in impoundments, although the slight differences may not be significant (Fig. 51). The relative response rates, as indicated by the parallel lines, are equal. The exact reasons for the apparent differences are unknown, but may be related to the generally shallower and hence warmer epilimnion of impoundments (Schindler 1971*b*). Natural lakes tend to be deeper and have larger epilimnetic volumes, resulting in cooler temperatures. Even though Hern et al. (1981) reported that temperature was not a major factor in the response of 757 National Eutrophication Survey lakes, temperature is known to influence phytoplankton succession and dominance (Fogg et al. 1973, Porter 1977).

The fact that depth and volume of the epilimnion may be an important factor in the chlorophyll *a*-total phosphorus correlation is further indicated by a comparison of stratified and mixed lakes (Fig. 52). Regressions for mixed lakes have a steeper slope and a higher R^2 value (41%) than stratified lakes (R^2 = 19%), indicating that mixed lakes have a much higher relative chlorophyll *a* response rate and a generally higher absolute chlorophyll *a* response per total phosphorus unit. The regressions indicate that 100 µg/l of total phosphorus should "produce" a chlorophyll *a* concentration of 28 µg/l in mixed lakes, but only 18 µg/l in stratified lakes. These differences occur even though mixed lakes had generally higher color and turbidities than stratified lakes, which should have resulted in lower chlorophyll *a* production per unit total phosphorus. The same effect was noted in a comparison of drainage and seepage lakes; drainage lakes had higher relative chlorophyll a response rates than seepage lakes.

The fact that mixed and drainage lakes have higher absolute response rates signifies that a nutrient other than phosphorus, or some other physical or chemical factors, may be affecting chlorophyll *a* concentrations. Higher levels of available forms of nitrogen or phosphorus or various micronutrients resulting from faster recycling or greater and continued influx of these essential elements in impoundments, mixed and drainage lakes may account for the relatively higher chlorophyll *a* production in these lake types. An alternative theory is that chlorophyll *a* measurements in impoundments, mixed lakes and drainage lakes may be slightly biased (overestimated) as a result of higher pheophytin levels (chlorophyll *a* degradation products) which may exist because of continued mixing and recycling of organic matter (see Moss 1973).

Nitrogen limitation and algal species composition are considered factors

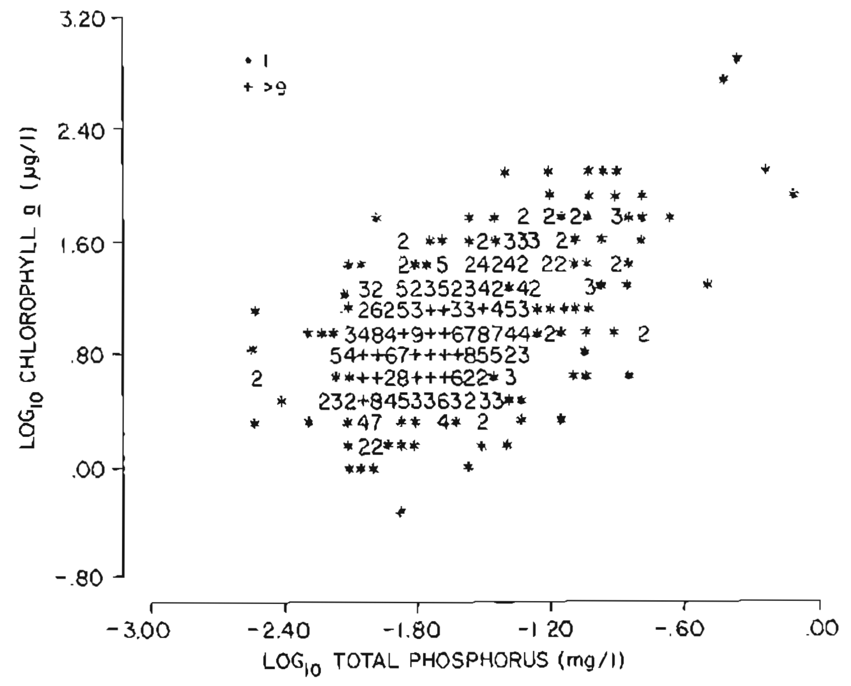


FIGURE 46. *Relationship of Log-transformed chlorophyll* a *and total phosphorus data for 641 randomly sampled Wisconsin lakes.*

TABLE 26. *Linear regression equations for log-transformed chlorophyll a and total phosphorus data based on lakes in the random data set.*

Log_{10} Chlorophyll *a* (µg/l) for:	=	Intercept	+	Slope Log_{10} Total Phosphorus (mg/l)	R^2	No.
All lakes		2.04	+	0.662	32.1%	641
N:P <10		2.11	+	0.882	51.3%	26
N:P 10-15		2.58	+	1.180	62.8%	30
N:P 15-25		2.44	+	0.964	43.6%	117
N:P >25		2.16	+	0.714	23.3%	466
Impoundments		2.02	+	0.622	38.2%	99
N:P <10		2.01	+	0.686	38.6%	13
N:P 15-40		2.23	+	0.746	24.6%	47
N:P 10-15		2.14	+	0.753	8.0%	12
N:P >40		1.70	+	0.456	9.0%	27
Lakes only		2.01	+	0.652	26.3%	539
N:P <10		2.02	+	0.873	28.5%	13
N:P 15-40		2.42	+	0.933	39.3%	252
N:P 10-15		2.70	+	1.280	62.7%	18
Stratified lakes and impds.		1.80	+	0.548	19.1%	277
N:P 15-40		2.20	+	0.826	32.8%	127
N:P >40		2.10	+	0.659	12.2%	129
Mixed lakes and impds.		2.12	+	0.680	41.0%	248
N:P < 15		2.21	+	0.889	35.7%	33
N:P 15-40		2.49	+	0.920	40.2%	113
N:P > 40		2.46	+	0.842	33.6%	101
Blue or Clear		1.02	+	0.190	3.0%	169
Green		2.46	+	0.828	51.7%	112
Brown		1.42	+	0.275	5.8%	204
Northeast		1.78	+	0.522	14.2%	235
Northwest		1.86	+	0.566	22.3%	272
Central		1.51	+	0.395	20.1%	43
Southeast		2.43	+	0.915	57.7%	61
Southwest		1.82	+	0.347	7.8%	29
Tot. phosphorus < 0.010 mg/l		0.79	+	0.044	0 %	137

possibly important in affecting chlorophyll *a* levels. Various ratios of nitrogen to phosphorus in lake waters have been suggested as limiting to phytoplankton growth (Gerloff and Skoog 1957, Uttormark, Chapin, and Green 1974, Lambou et al. 1976, Claesson and Ryding 1977, Schindler 1978). These values range from 5:1 to 15:1, but in general, lakes with ratios above 15:1 have been considered to be phosphorus-limited. Ratios between 10-15:1 are considered to be transitionary and values below 10:1 are generally thought to indicate nitrogen limitation.

Figure 53 shows the chlorophyll *a*-total phosphorus relationship for the same sets of lakes as presented in Figure 46, except lakes were coded as to their N:P ratios. The impact of the N:P ratio on the chlorophyll *a*-total phosphorus relationship is readily apparent and is further depicted in Figure 54 and Table 26. The R^2 value is highest for lakes with N:P less than 15:1. The relationship is weakest for lakes with N:P 15-25, which, incidentally, has a slope quite similar to that shown for the three cited references. Similar plots for impoundments and natural lakes (Figs. 55, 56) indicate that the impact of the N:P ratio is less significant in the case of impoundments where slopes of the regression lines remain essentially parallel. However, R^2 values for natural lakes with N:P ratios from 10-40 are much higher than for impoundments with similar N:P ratios. Lakes and impoundments with N:P ratios from 15-40 had somewhat similar slopes. The lower R^2 values and slopes for lakes and impoundments with N:P ratios greater than 40:1 are believed to be associated with the inclusion of a disproportionately large number of lakes with low total phosphorus (78%

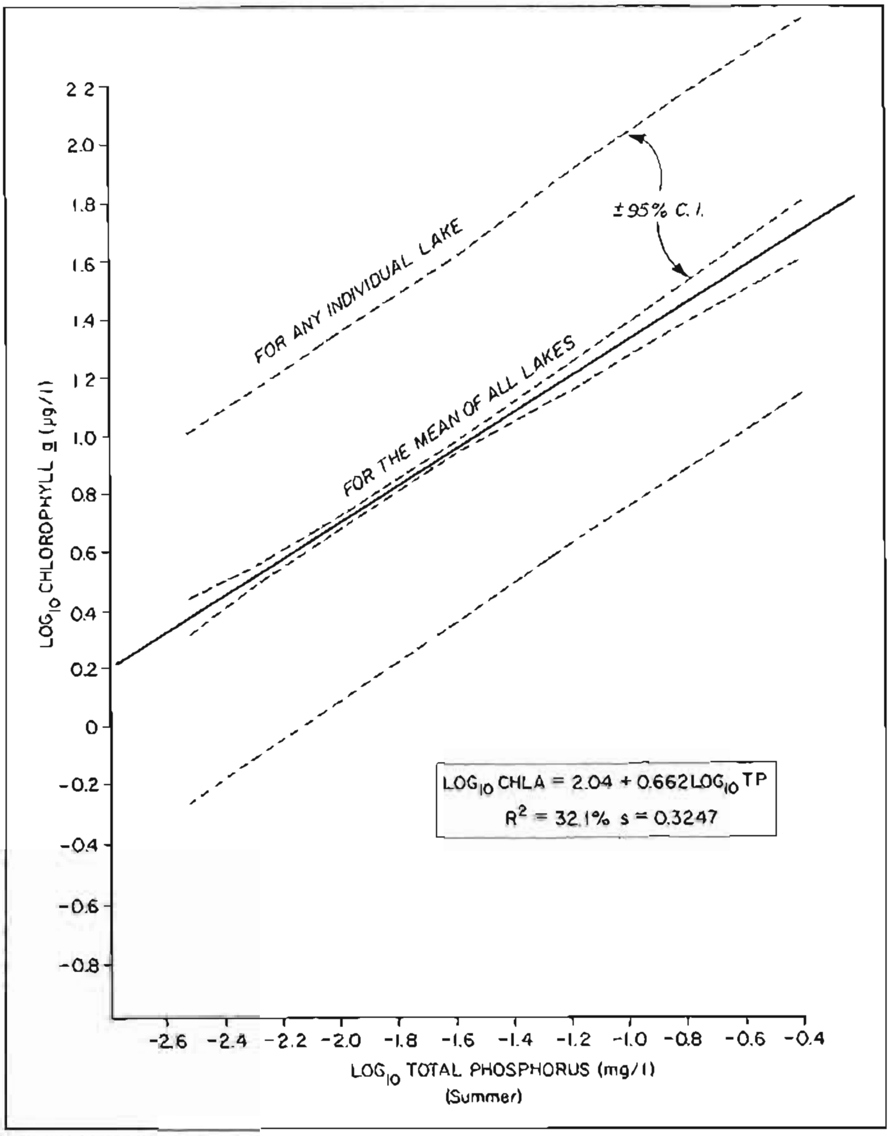


FIGURE 47. *Confidence interval bands for predictive purposes using the Log chlorophyll* a*-Log total phosphorus relationship for 641 randomly sampled Wisconsin lakes.*

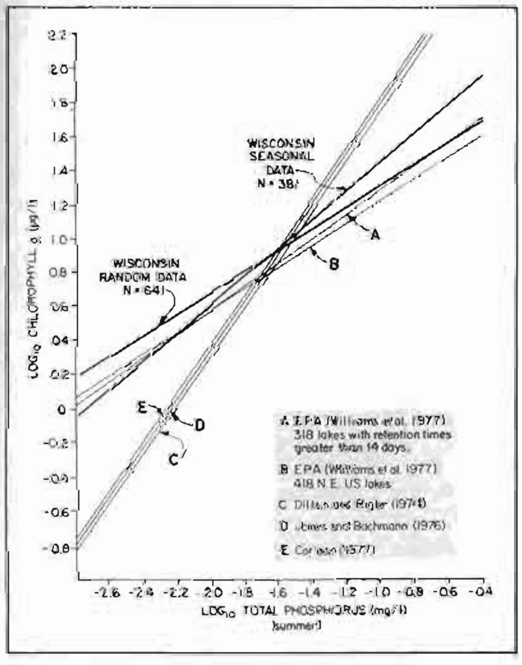
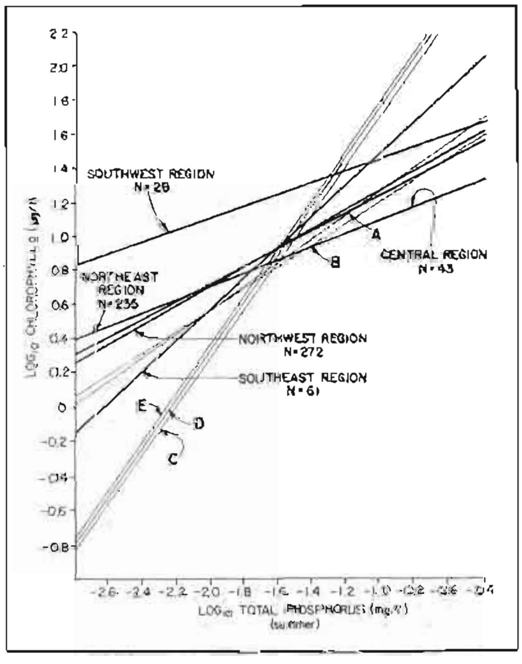
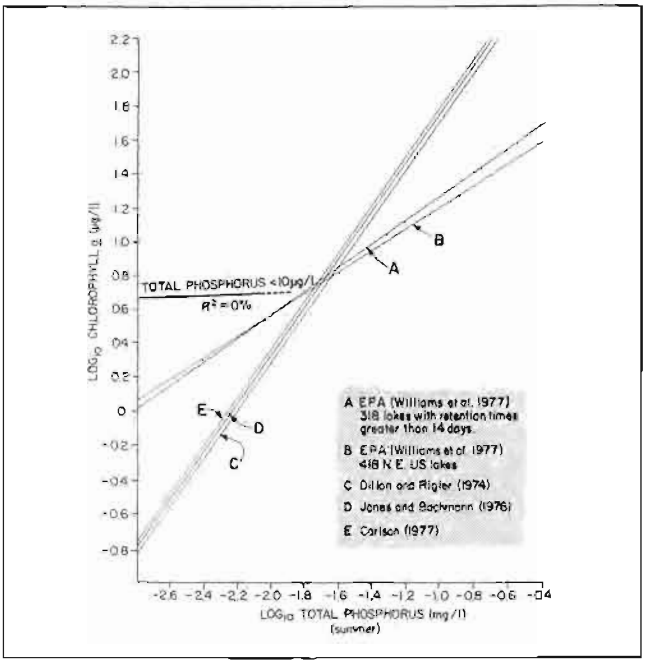
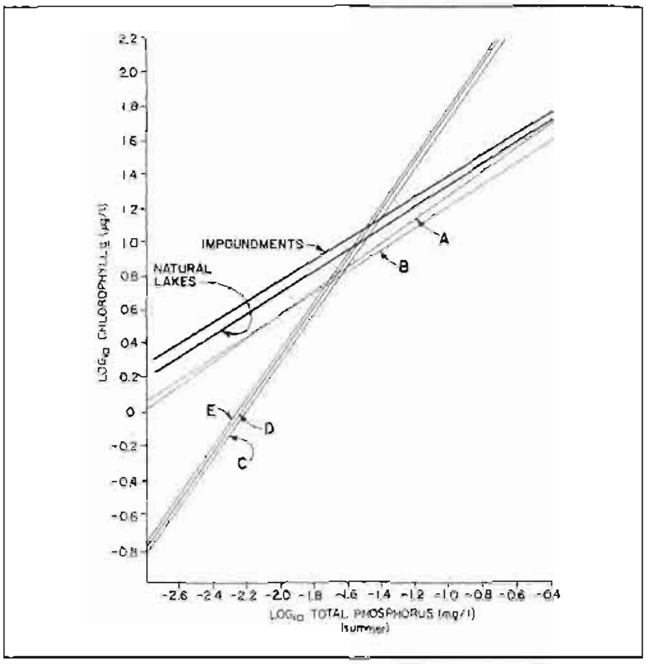


FIGURE 48. *Comparison of Log-transformed chlorophyll* a *-total phosphorus relationship for Wisconsin lake data and other referenced data sets. (left)*

FIGURE 49. *Regional comparisons of the chlorophyll* a *-total phosphorus relationship. (right)*







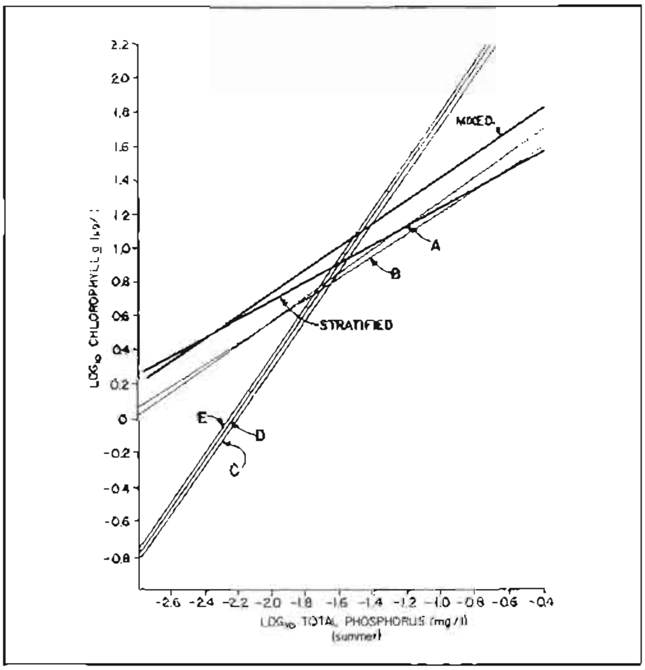


FIGURE 50. *Relationship between chlorophyll* a *and total phosphorus for lakes with total phosphorus levels less or greater than 10 µg/l. (top left)*

FIGURE 51. *Chlorophyll* a *-total phosphorus relationship for natural lakes and impoundments. (top right)*

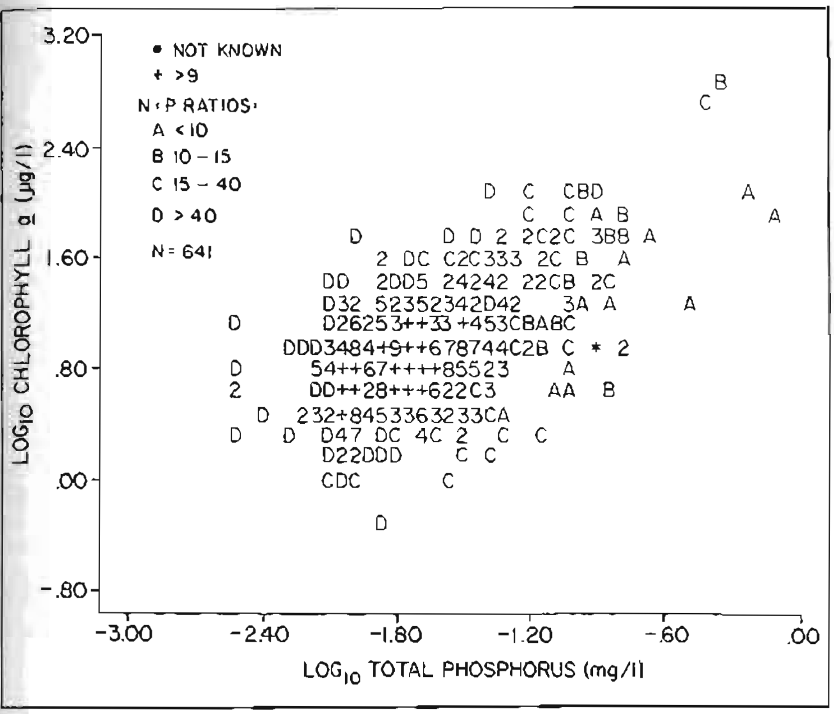
FIGURE 52. *Chlorophyll* a *-total phosphorus relationship for mixed lakes and impoundments and stratified lakes and impoundments. (bottom left)*

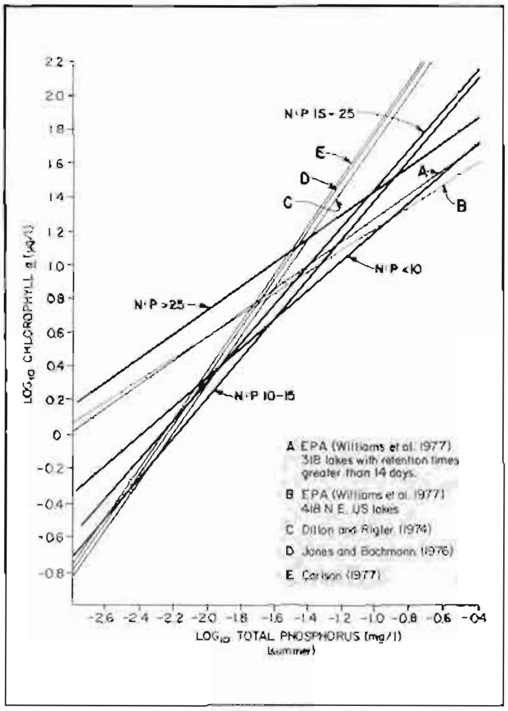
had total phosphorus levels less than 15 µg/l). Natural lakes with N:P ratios from 10-15 had the highest R^2 (62.7%) and also had a slope that was nearly identical to the reference plots. The R^2 for impoundments, within the same N:P limitations, was very poor, and yet the regression was similar to the EPA-NES results (Williams et al. 1977).

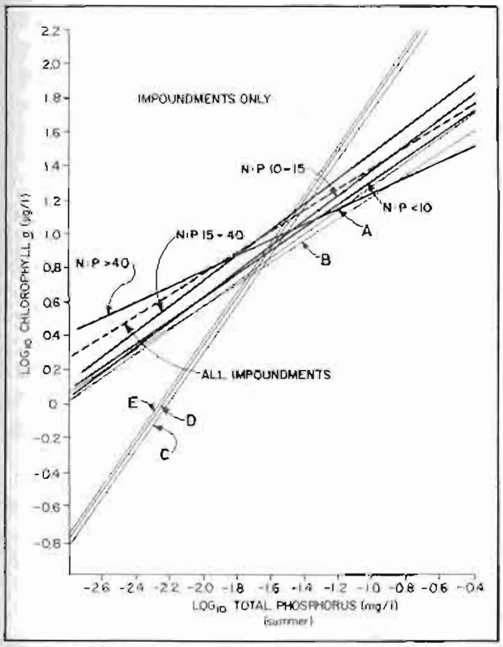
The impact of the N:P ratio on the chlorophyll *a*-total phosphorus relationship varies with lake type and appears to indicate (for natural lakes at least) that the transition zone is indeed in the 10-15:1 region. The relative chlorophyll *a* response rate for lakes with N:P less than 10:1 is somewhat parallel to those lakes with N:P from 15-40 and is much steeper for the lakes with N:P from 10-15. The absolute chlorophyll *a* response is much lower for the lakes with N:P less than 10:1 (supporting the nitrogen-limitation theory) than in the other two groups of lakes (as will be shown later, most of the lakes with very low N:P ratios have total phosphorus levels exceeding 30 µg/l).

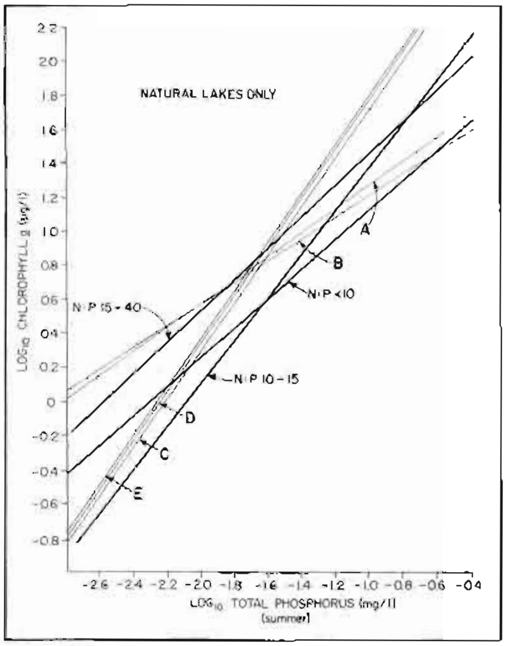
While the N:P ratio has a significant impact on the total phosphorus-chlorophyll *a* and total phosphorus-water clarity relationships (Figs. 53 and 57), it does not dramatically affect the water clarity-chlorophyll *a* relationship.

It is evident that considerable variation exists in the chlorophyll a-total phosphorus relationship, and while some of this variation may be due to nitrogen limitation, apparently other factors must also influence the relationship. What these factors might be is not known. Resuspension of dead phytoplankton or the contribution of littoral zone phytoplankton may possibly help account for the generally higher chlorophyll *a* levels associated with mixed lakes and impoundments (Moss 1973). Algal species composition may also play an important role. Bush (1971) reported that variations in biomass (chlorophyll *a*) in a eutrophic Washington lake were best explained by inorganic phosphorus when dominated by green or bluegreen algae and by nitrate nitrogen when diatoms dominated; even when the lake appeared to be nitrogen limited, inorganic phosphorus levels best explained the variations in biomass when bluegreen algae









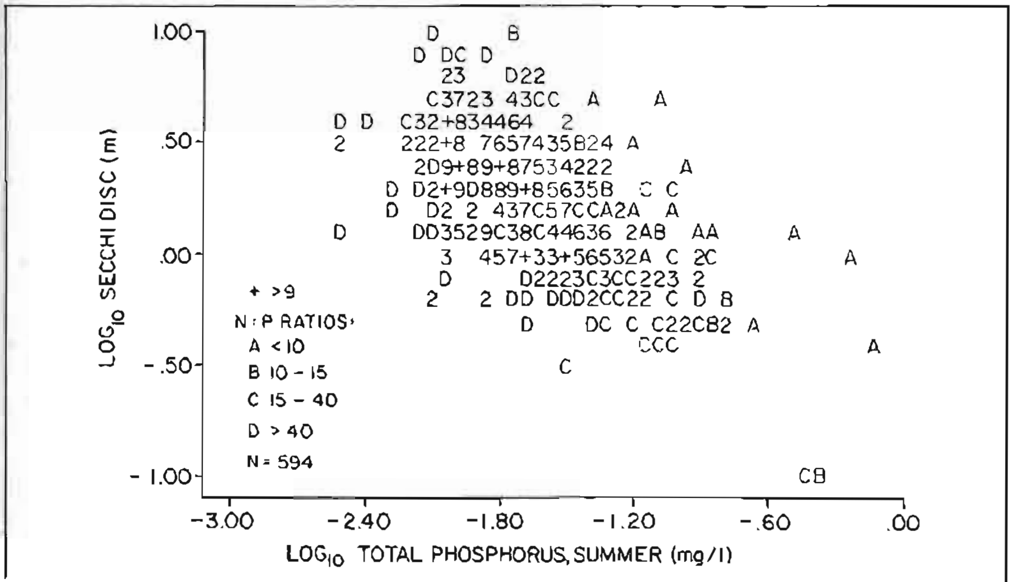


FIGURE 53. *Relationship of Log-transformed chlorophyll* a *and total phosphorus data for 641 Wisconsin lakes. (top left)*

FIGURE 54. *Impact of differing total nitrogen-to-total phosphorus ratios on the chlorophyll* a *-total phosphorus relationship in the randomly sampled Wisconsin lakes. (top right)*

FIGURE 55. *Impact of nitrogen-to-phosphorus ratios on the total phosphorus-chlorophyll* a *relationship among the impoundments of the random data set. (middle left)*

FIGURE 56. *Impact of nitrogen-to-phosphorus ratios on the total phosphorus-chlorophyll* a *relationship among the natural lakes of the random data set. (middle right)*

FIGURE 57. *Relationship of water clarity and total phosphorus for 594 randomly sampled Wisconsin lakes. (bottom)*

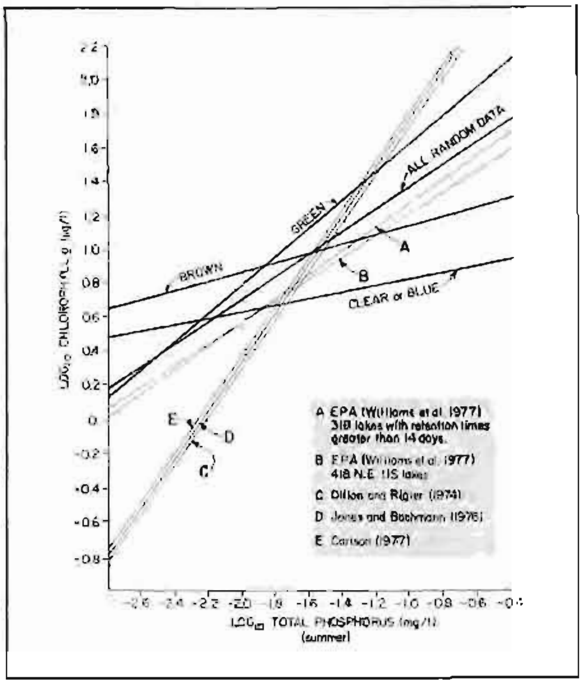


FIGURE 58. *Relationship of chlorophyll* a *and total phosphorus for lakes with different perceived color.*

dominated. Phytoplankton samples collected during the 1979 random survey have not been analyzed to determine if these findings may be valid for Wisconsin lakes.

Theoretically, light limitation caused by turbidity, color, or self-shading by algae would be expected to lower the absolute chlorophyll *a* response rates and decrease the R^2 values of the regressions. However, analysis of the differences between seepage and drainage, stratified and mixed, and impoundments and natural lakes indicates that lakes with higher turbidities and/or measured color generally tended to have higher R^2 values and absolute chlorophyll a responses. Whether this was due to greater productivity rates created in part by the generally warmer, shallower conditions found in these lakes, or due to other unidentified factors, is uncertain.

Color does appear to be very important. The chlorophyll *a*-total phosphorus relationship differs for lakes with different perceived water color (Fig. 58). The R^2 values and response rates for brown and clear lakes were very poor, while green lakes had a higher relative response rate and high R^2 (51.7%). The poor relative response rate indicated for clear lakes may be associated with the large number of lakes with low total phosphorus levels (Fig. 50) and suggests that these lakes produce less chlorophyll *a* per unit total phosphorus than other lakes. This may be related to the luxury uptake of phosphorus within a relatively few cells per liter. The brown lakes also show a very poor relationship, but the regression line is parallel and above that shown for clear lakes. This may be an artifact of the distribution of total phosphorus and chlorophyll *a* values in the two data sets. The brown lakes have a lower absolute response of chlorophyll *a* to total phosphorus than green lakes above a total phosphorus concentration of 0.013 mg/l, which is what might be expected if color were inhibiting chlorophyll *a* production. It appears that factors other than color may explain the greater chlorophyll *a* response per unit total phosphorus in mixed lakes, impoundments and drainage lakes.

Further Considerations

As we have demonstrated, various levels of chlorophyll *a*, total phosphorus and water clarity can be suggested as indicative of different lake trophic status and overall lake water quality (Table 27). We have shown that trophic status classifications are based on somewhat arbitrary divisions of perceived water quality or on relationships with other parameters. Generally, total phosphorus values of 10 µg/l and 20 µg/l have been used in previous reports for separating lakes into different trophic classes (Bartsch and Gakstatter 1978, Hern et al. 1981). Using these total phosphorus values as a guide, corresponding chlorophyll *a* values for separating Wisconsin lakes according to trophic class would be 5.2 µg/l and 8.2 µg/l, respectively (5.1 µg/l and 8.0 µg/l for natural lakes only).

However, because of the great variability in the total phosphorus-chlorophyll *a* relationship in the low ranges, the chlorophyll a-water clarity relationship probably gives a better representation of the actual trophic status of Wisconsin lakes. Based on water clarity criteria equating 2.0 and 3.7 m as trophic status division lines (U.S. EPA 1975), the corresponding chlorophyll *a* levels based on our random data set would be 2.3 µg/l and 7.7 µg/l, respectively (2.5 µg/l and 8.3 µg/l for natural lakes; 2.9 µg/l and 9.6 µg/l for seepage lakes) (Table 28). These values are very similar to those suggested by Hern et al. (1981) (2.3 µg/l and 6.4 µg/l chlorophyll *a*).

Criteria have been established by which summer trophic state or water quality can be predicted based on springtime concentrations of nutrients. Sawyer (1947) proposed spring inorganic nitrogen and inorganic phosphorus levels of 0.30 mg/l and 0.01 mg/l as minimum requirements for the development of summer algae blooms. Only 28% of the lakes in the quarterly data base had spring inorganic nutrient concentrations less than this level, which might be expected considering the bias in the quarterly data set towards eutrophic lakes. A comparison of mean chlorophyll *a* and nutrient data (quarterly data set) for lakes and impoundments with less than 0.30 mg/l inorganic nitrogen and 0.01 mg/l inorganic phosphorus illustrates that Sawyer's standards are most applicable to stratified natural lakes (Table 29). If 10 µg/l chlorophyll *a* is selected as an indicator level of visible blooms (based on our observations that indicate clear lakes generally have chlorophyll *a* levels <10 µg/l), then stratified natural lakes have average chlorophyll *a* levels (7.7 µg/l) that are almost invisible. Seventy-seven percent of the stratified natural lakes with low inorganic nutrient levels had summer chlorophyll *a* levels less than 10 µg/l (Table 30). Impoundments on the average were found to have clearly visible algae blooms in summer, based on mean chlorophyll *a* contents of 21.9-26.1 µg/l. Total phosphorus levels in impoundments and mixed natural lakes showed significant increases from spring to summer, while total phosphorus in stratified lakes showed only a slight increase (summer mean = 0.023 mg/l). Therefore, it appears that Sawyer's numbers are applicable only to natural lakes in Wisconsin. In impoundments, high flow-through rates (low retention times), seasonally variable nutrient loadings, nutrient recycling, macrophyte growth, and other factors tend to cause greater nutrient fluxes, and as a result spring conditions do not necessarily serve as a basis for predicting summer in-lake conditions.

The data presented in Tables 29 and 30 for Wisconsin lakes support the phosphorus guidelines developed by other investigators which state that in-lake total phosphorus concentrations in excess of 20 µg/l during spring turno-

TABLE 27. *Trophic classification values for chlorophyll a reported in the literature.*

	Sakamoto (1966)	Nat. Acad. Sci (1972)	Dobson et al. (1974)	Bartsch and Gakstatter (1978) Lake Survey EPA-NES 1978	Daws (1978), Likens (1975)	Hern et al. (1981) (EPA)	Chapra and Tarapchak (1976), R.E. Carlson (1975 unpubl.)
Oligotrophic	0.3-2.5*	0-4	0-4.4	0-7	0-3	0-2.3	1
Mesotrophic	1-15	4-10	4.4-8.8	7-12	2-15	2.3-6.4	1-6
Eutrophic	5-140	10	8.8	12	10-500	6.4	6

*Chla values in μg/l.

TABLE 28. *Chlorophyll* a *and water clarity measurements associated with specific total phosphorus values used in trophic classification.*

	Chlorophyll a (μg/l)			Secchi Disc (m)	
Total Phosphorus (μg/l)	Bartsch and Gakstatter (1978)	Hern et al. (1981)	Wisconsin Random Sample Data	(Wisconsin Random Sample Data)	Trophic Class (Vollenweider 1968)
3			2.3	3.7	Oligotrophic
10	7	2.3	5.2	2.4	
18	12		7.7	2.0	Mesotrophic
20		6.4	8.2	1.9	
27			10.0*	1.8	
30			10.7	1.7	Eutrophic
40					
50			15.0	1.4	
60					

* 87% of lakes with total phosphorus <10 μg/l had chlorophyll *a* <10 μg/l. Vallentyne (1969) suggested that lakes with chlorophyll *a* levels exceeding 10 μg/l may have algal blooms sufficient to impair recreational activities.

TABLE 29. *Comparison of characteristics of lakes and impoundments with low spring inorganic nutrient levels (quarterly data set).*

	Lakes With < 0.3 mg/l Spring Inorganic Nitrogen and < 0.01 mg/l Spring Inorganic Phosphorus																		
	All Lakes In Data Set			Impoundments						Natural Lakes									
				Mixed			Stratified			Mixed			Stratified			All			
	N	$\bar{x}$	SD	N	$\bar{x}$	SD	N	$\bar{x}$	SD	N	$\bar{x}$	SD	N	$\bar{x}$	SD	N	$\bar{x}$	SD	Median
Inorganic P, spring	507	0.026	0.044	7	0.009	0.001	2	0.008	0.003	43	0.008	0.002	80	0.006	0.002	153	0.007	0.002	0.007
Inorganic N, spring	507	0.470	0.52	7	0.15	0.07	2	0.18	0.01	43	0.17	0.06	80	0.13	0.06	153	0.15	0.07	0.14
Organic N, spring	507	0.551	0.271	7	0.40	0.14	2	0.61	0.12	43	0.50	0.18	80	0.38	0.14	153	0.42	0.17	0.40
Total N, spring	507	1.022	0.688	7	0.55	0.18	2	0.79	0.13	43	0.68	0.20	80	0.51	0.16	153	0.58	0.20	0.58
Total P, spring	539	0.061	0.062	7	0.031	0.022	2	0.035	0.000	43	0.025	0.016	80	0.020	0.010	153	0.023	0.013	0.020
Total P, summer	507	0.061	0.094	7	0.043	0.024	2	0.046	0.013	41	0.036	0.024	78	0.025	0.013	148	0.030	0.021	0.025
Chlorophyll a, summer	381	28.8	65.9	7	26.1	15.9	2	21.9	13.3	34	18.8	19.1	53	7.7	7.2	113	13.9	14.9	8.1
Drainage Basin: lake area.	522	166	592	7	82	81	2	9	3	41	28	47	78	14	33	148	33	98	6
Retention time	430	1.49	1.80	5	0.32	0.22	2	0.76	0.05	32	1.14	0.93	58	3.42	2.65	114	2.25	2.31	1.68

TABLE 30. *Comparison of percentages of lakes with low inorganic nutrient levels (<0.3 mg/l inorganic nitrogen and <0.01 mg/l inorganic phosphorus) during spring turnover in relation to summer total phosphorus and chlorophyll* a *levels and spring total phosphorus levels for different lake types.*

	Summer Chlorophyll *a* (μg/l)				Summer Total P Concentration (mg/l)				Spring Total P Concentration (mg/l)			
	<5	<10	<15	<25	<0.01	<0.02	<0.03	<0.05	<0.01	<0.02	<0.03	<0.05
Impoundments												
Mixed	14	28	28	43	0	29	43	57	0	14	29	43
Stratified	0	0	50	50	0	0	0	50	0	0	0	100
Natural Lakes												
Mixed	15	50	62	80	10	34	56	80	12	44	74	88
Stratified	49	77	90	96	12	54	76	92	14	62	87	95

ver would likely produce chlorophyll *a* concentrations over 10 µg/l during the summer (Vollenweider 1968, Dillon 1975). These concentrations appear to be applicable to other seasons as well. Bachmann and Jones (1974) found that a chlorophyll *a* value of 7.5 µg/l corresponded to an annual median total phosphorus concentration of 20 µg/l. This corresponds well with the 7.7 µg/l mean chlorophyll *a* concentration for all stratified natural lakes (Table 29) with spring total phosphorus concentrations averaging 20 µg/l and summer total phosphorus concentrations averaging 23 µg/l. Bachmann and Jones suggested that a reduction in total annual phosphorus concentrations below 20 µg/l would result in significant improvements in water clarity. Among Wisconsin stratified natural lakes with low spring concentrations of nutrients, a median summer total phosphorus value of 20 µg/l corresponded to a median chlorophyll *a* concentration of 5.2 µg/l (Table 31). Overall, our data support Bartsch and Gakstatter's (1978) evaluation that total phosphorus levels should be maintained below 20-30 µg/l during any season to avoid nuisance algae blooms.

TABLE 31. *Summer chlorophyll a and total phosphorus levels (medians) in lakes with low inorganic nutrient levels (<0.3 mg/l inorganic nitrogen and <0.01 mg/l inorganic phosphorus) during the spring sampling period.*

	Impoundments		Natural Lakes	
	Mixed	Stratified	Mixed	Stratified
Chlorophyll *a* medians (µg/l)	34.4	21.9	11.6	5.2
Total P medians (mg/l)	0.045	0.046	0.030	0.020

The Southeastern Wisconsin Regional Planning Commission has chosen 20 µg/l as a lake water quality standard. Our statewide data suggest that 30 µg/l may be a reasonable secondary standard for good water quality, which should assure relatively clear water in the majority of cases.

Evidence in support of a 30 µg/l total phosphorus level includes the fact that the mean summer total phosphorus concentration of stratified natural lakes in the quarterly data set (23 µg/l) corresponded with a mean summer chlorophyll *a* concentration (7.7 µg/l) which was less than visible (Table 29) and the fact that 53% of the lakes with total phosphorus levels exceeding 30 µg/l had chlorophyll *a* levels less than 15 µg/l (Table 32). Fifty-five percent of the lakes with chlorophyll *a* levels exceeding 15 µg/l also had summer total phosphorus levels greater than 30 µg/l. Eighty-three percent of the lakes with total phosphorus less than 10 µg/l had water clarity greater than 2 m, while 78% of the lakes with total phosphorus greater than 30 µg/l had water clarity less than 2 m. The majority of Wisconsin's softwater lakes (alkalinity less than 30 mg/l) had total phosphorus less than 30 µg/l).

The major difficulties encountered in establishing uniform cutoff points for classifying lakes according to trophic status based on the water clarity-chlorophyll *a*-total phosphorus relationships are that the interrelationships between these three parameters are not consistent and the relationships are affected by the natural daily, seasonal and annual variations that occur in lakes.

For Wisconsin lakes, the relationship between water clarity and total phosphorus has been found to be far from linear or constant; changes in total phosphorus levels are not always accompanied by corresponding changes in water clarity or chlorophyll *a*. Figure 59 gives an example of the variations or cycles in the total phosphorus-water clarity relationship that occurred in one lake in southeastern Wisconsin; the pattern is known to be different in other lakes. Some lakes demonstrate a counterclockwise rotation in the early portion of the year, and some experience much larger fluctuations during the summer months. The fact that these changes seem to follow a repetitive pattern from one year to the next and display a somewhat logical overall pattern seems to indicate that laboratory precision was not a major factor in the variations observed.

It appears that two lakes may have the same total phosphorus concentrations and yet have considerably different chlorophyll *a* concentrations and/or water clarity due to various factors discussed previously such as: (1) low N:P ratios, (2) light limitation caused by high inorganic turbidities or color, (3) inhibition of chlorophyll *a* production due to toxic substances, or (4) differences in biological interactions. One of the most important factors may be the difference in biological communities and interrelationships found in individual lakes. Important biological considerations may include the effects of herbivorous zooplankton, high grazing rates, and the release of soluble phosphorus and nitrogen by both zooplankton and phytoplankton. Crashes in phytoplankton populations can and do

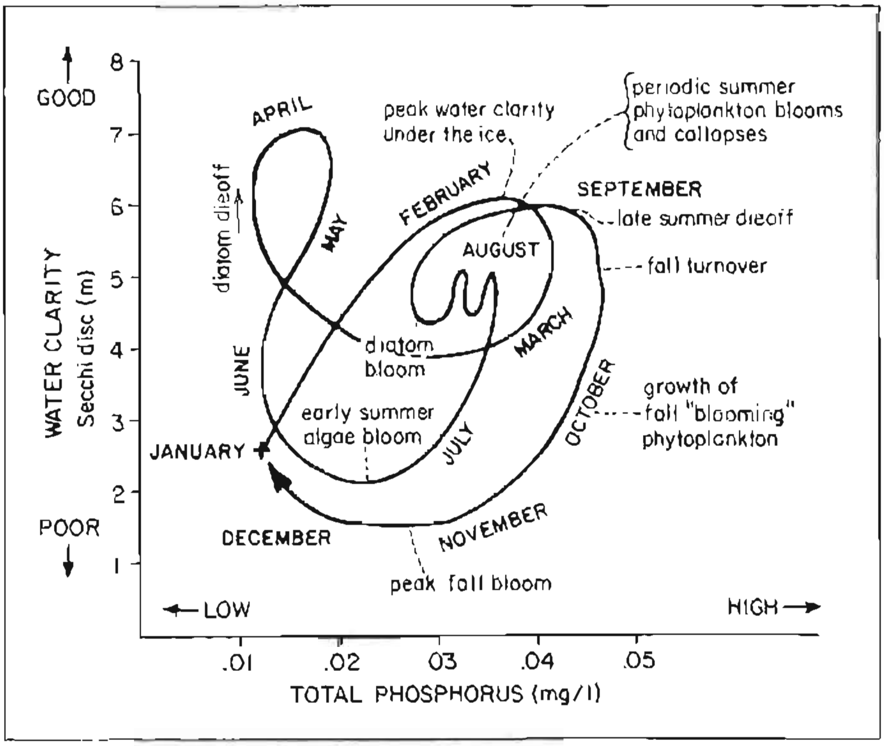


FIGURE 59. *Schematic diagram illustrating the cyclic relationship between total phosphorus and water clarity in Oconomowoc Lake, Waukesha County, Wisconsin, during 1976-77. Data represents 20 separate sampling dates, 1976-77. Source: Wis. DNR and Southeast Wis. Reg. Plan. Comm.*

TABLE 32. *Distribution of lakes within various ranges of chlorophyll a, water clarity, total alkalinity and total phosphorus (random data set).*

		Total Phosphorus (Summer)									
		0-10 µg/l			10-30 µg/l			> 30 µg/l			
		Lakes		% with	Lakes		% with	Lakes		% with	Total
	Range	No.	%	Low P	No.	%	Med. P	No.	%	High P	Lakes
	0-5	70	51	37	101	29	53	19	12	10	190
	5-10	50	36	21	144	41	61	44	27	18	238
Chlorophyll *a*	10-15	7	5	8	53	15	63	24	15	29	84
(µg/l)	15-30	9	7	11	42	12	49	35	21	41	86
	> 30	1	1	2	11	3	21	41	25	77	53
	Total	137			351			163			651
	0-1	5	4	5	32	10	32	62	39	62	99
Secchi disc	1-2	16	13	9	106	33	58	61	39	33	183
(m)	2-3	37	30	22	105	32	64	23	14	14	165
	3-4	33	27	35	48	15	51	13	8	14	94
	4-5	20	16	50	20	6	50	0	-	-	40
	5-6	6	5	40	7	2	47	2	1	13	15
	> 6	5	5	42	7	2	58	0	-	-	12
	Total	122			325			161			608
	0-10	45	32	25	115	32	64	21	12	12	181
Total	10-30	20	14	13	92	26	61	39	23	26	151
alkalinity	30-90	36	26	19	92	26	48	65	38	34	193
(mg/l)	> 90	39	28	27	61	17	42	45	26	31	145
	Total	140			360			170			670*

*A few lakes were included in two sections.

occur frequently and could greatly affect interrelationships.

A schematic illustration of the effects of different variables on the water clarity-total phosphorus-chlorophyll *a* relationship is presented in Figure 60. Lakes may receive different classifications depending upon which of the parameters is chosen as a trophic status indicator. While color and turbidity corrections or modifications to the water clarity-chlorophyll *a* relationship have been proposed (Garn and Parrott 1977, Brezonik 1978), these corrections would not take into account the probable increase in the chlorophyll *a* response rate which might occur if light-limiting interferences were removed. In this respect, the correction would tend to overcompensate for the interferences in the water clarity measurement. For example, a lake with a chlorophyll *a* of 2.3 µg/l, water clarity of 2.0 m, and total phosphorus concentration of 15 µg/l adjusted for color and turbidity which caused the low water clarity condition (position A, Figure 60), would shift the lake to position B. This would not necessarily be accurate, because the chlorophyll *a* response may be presently inhibited by the high color or turbidities.

A better representation of what might occur would be an increase in chlorophyll *a* parallel to the total phosphorus axis (Fig. 60, position C). This would best represent the estimated increase in chlorophyll *a* production if color and turbidity were reduced while total phosphorus stayed the same. However, this may not necessarily be the case either, as some of the total phosphorus may not be biologically available (Schaffner and Oglesby 1978), at least not in direct proportion to the amount of color or turbidity interference. Nevertheless, an adjustment which takes into account the total phosphorus potentially available for chlorophyll *a* production seems to be more realistic than a simple correction for color or turbidity.

Another factor, which is evident in Figure 44, is the effect the N:P ratio, or possibly nitrogen limitation, has on these relationships. Lakes with low N:P ratios have lower chlorophyll *a* production and better water clarity than would be expected based on the amount of total phosphorus available (e.g., Fig. 60, position D). In the theoretical case shown, the lake has no interferences due to either color or turbidity, and trophic classification according to water clarity and chlorophyll *a* agree. However, based on total phosphorus levels the lake should be classed in the eutrophic category (position E); therefore, nitrogen limitation or other factors which reduce chlorophyll *a* response may lead to an underestimation of the potential trophic condition of some lakes.

As a further example of the potential significance of the N:P ratio, a hypothetical case is proposed in which lake A (Fig. 60) is also nitrogen limited. A small increase in nitrogen may shift the lake to the eutrophic class (position F). An unknown in this instance would be whether or not this shift from nitrogen-limited to phosphorus-limited conditions would be accompanied by further deterioration of the lake's water quality. Preliminary studies indicate the possibility that such a change might cause a shift from noxious blue-green algal communities to green algae, which are generally acknowledged to be more highly utilized by zooplankton. Such changes could potentially result in increased fish production and improved lake water quality. The potential improvement in actual or perceived water quality may have great impact on the perceived trophic or water quality characteristics of a lake.

Summary

The classification of lakes according to their trophic status and the assessment of water quality rely heavily upon the assumed relationships between water clarity, total phosphorus and chlorophyll *a* concentration. There is variability in these relationships and many factors influence them. For classifying lakes on the basis of water clarity, total phosphorus content, or chlorophyll *a* level, the selection of the single best indicator would have to be based on such considerations as (1) whether or not the actual in-lake biological production as expressed by chlorophyll *a* biomass is more informative, or (2) whether the potential production based on total phosphorus values and computed (predicted) production is

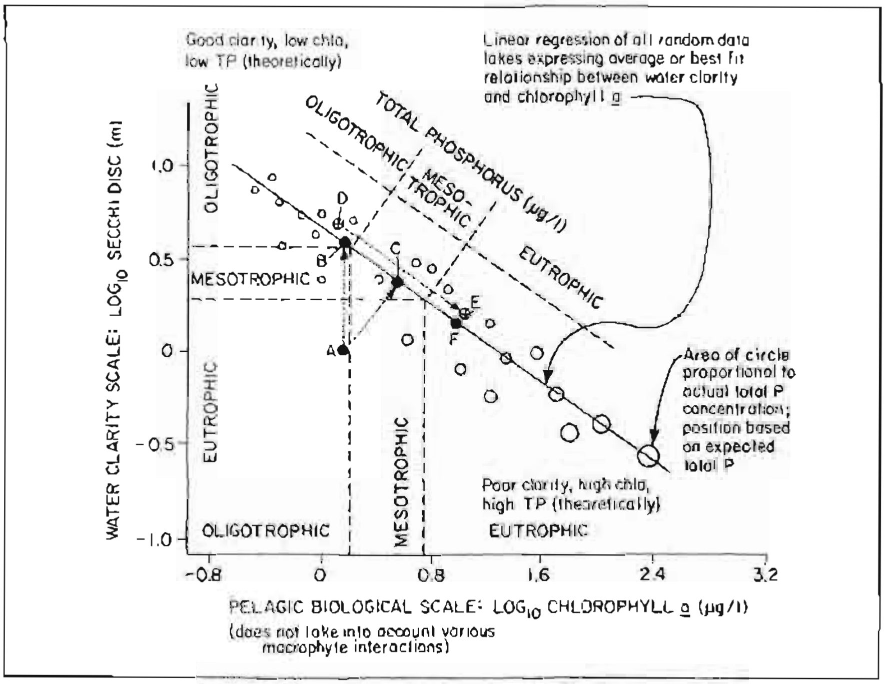


FIGURE 60. *Generalized schematic diagram expressing the interdependent relationships between water clarity, biological production (biomass, in this case measured by chlorophyll* a*), and total phosphorus. (See text for explanation of letters.)*

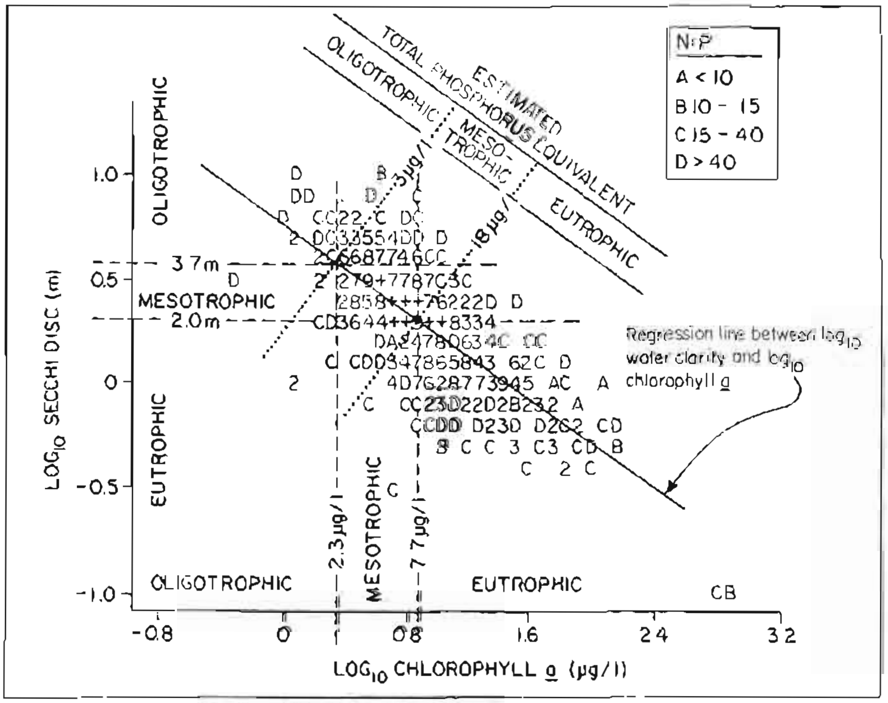


FIGURE 61. *Trophic classification of 578 Wisconsin lakes based on water clarity, chlorophyll* a*, or total phosphorus concentrations (from data in Table 27 and linear regressions provided in Tables 23 and 25, representing all random data).*

more relevant. It should be reiterated that the biota of any lake is extremely important in the dynamic processes occurring within the lake, and that subtle changes in biota may result in drastic changes in the perceived trophic state (Shapiro 1978, Barica 1974).

Many Wisconsin lakes could be placed in different trophic status classes depending on the parameter selected as the basis for classification. Figure 61 shows 578 randomly sampled lakes in relation to the log-transformed water clarity-chlorophyll a relationship; superimposed on this plot are lines representing different trophic "cutoff" values and a third scale representing equivalent lines of total phosphorus based on the linear regression between total phosphorus-chlorophyll *a* for the random data set. It must be emphasized that the total phosphorus scale represents the *means* for all lakes with a particular chlorophyll *a* level and that many lakes with chlorophyll *a* levels and water clarity levels as shown will have considerably higher levels of total phosphorus (see Fig. 44 for actual examples). For these reasons, phosphorus level alone appears to be a poor indicator of a lake's trophic status.

It is apparent from Figure 61 that considerable error may result from the use of a single parameter to classify lakes according to their trophic status. Regardless of how simple some trophic state index systems may seem, the accurate assessment and comparison of the trophic status of lakes remain quite complex and open to question. If a lake must be classified on the basis of only one trophic indicator, chlorophyll *a* concentrations (summer means or perhaps maxima) seem to be the best overall index values since they reflect the actual conversion (except in the case of macrophyte-dominated lakes) of nutrients to biomass. An alternative method, where practical, might be to record the number of days or the percentage of the summer period that certain water quality conditions exist (i.e., number of days a lake appeared green or chlorophyll *a* levels exceeded a particular level).

TABLE 33. *Comparison of summer nitrogen and phosphorus levels in Wisconsin lakes based on different data sets.*

	All Data				Random Data				Seasonal Data			
	No. Lakes*	Mean	SD**	CV**	No. Lakes	Mean	SD	CV	No. Lakes	Mean	SD	CV
Organic N	1287	0.624	0.380	61%	659	0.601	0.356	59%	539	0.653	0.398	61%
Total N	1287	0.911	0.637	70%	659	0.856	0.568	66%	539	0.971	0.701	72%
Inorganic P	1285	0.022	0.051	235%	658	0.013	0.036	271%	539	0.031	0.063	203%
Total P	1286	0.045	0.076	170%	659	0.031	0.050	164%	539	0.061	0.094	154%

*Some lakes were in both data sets; thus totals exceed 1,140 lakes.
**SD = standard deviation; CV = coefficient of variation.

Nutrient Concentrations

Nutrient concentrations in Wisconsin lakes are highly dependent on watershed characteristics and lake types. Although this study was not designed to measure or evaluate specific sources (point or non-point) of nutrient inputs into lakes, certain gross generalizations can be made concerning the relationship of certain watershed characteristics to in-lake water quality conditions. The associations of the plant nutrients, phosphorus and nitrogen, with other lake and watershed characteristics and the dynamic nature of these elements in Wisconsin lakes will be discussed in detail in this section.

Phosphorus dynamics

Phosphorus content of lakes in Wisconsin (epilimnion) was found to be highly variable (Table 33). Coefficients of variation were much higher for phosphorus parameters as opposed to the smaller values for nitrogen regardless of the data set. Seasonal variations in phosphorus concentration were somewhat different for different lake types (Figure 62). While seasonal phosphorus means (both inorganic and total) within similar lake types were not significantly different, the changes in the absolute sample means showed some consistencies.

A closer examination of only the total phosphorus means for six lake types demonstrates that mixed and stratified impoundments and mixed drainage lakes generally have higher levels of phosphorus than mixed or stratified seepage lakes or stratified drainage lakes, regardless of season (Fig. 63). Also important is the characteristic of stratified lakes to demonstrate a decrease in total phosphorus from spring to summer while mixed lakes show an increase (Fig. 63). The mean total phosphorus of stratified lakes reaches a maximum in the fall, which may have important implications in evaluating the trophic level of stratified lakes based on summer data only. There appears to be a rather consistent net increase in total phosphorus from spring to fall in both mixed and stratified seepage and drainage lakes that may represent the net accumulations of phosphorus in each of the systems over that period.

Inorganic phosphorus dynamics closely resemble those of total phosphorus (Fig. 62). No significant differences were found in the inorganic-organic fractions of phosphorus for different lake types when comparing the seasonal means for matched sets of mixed and stratified lakes (Fig. 65). However, there is some indication that impoundments have a generally lower

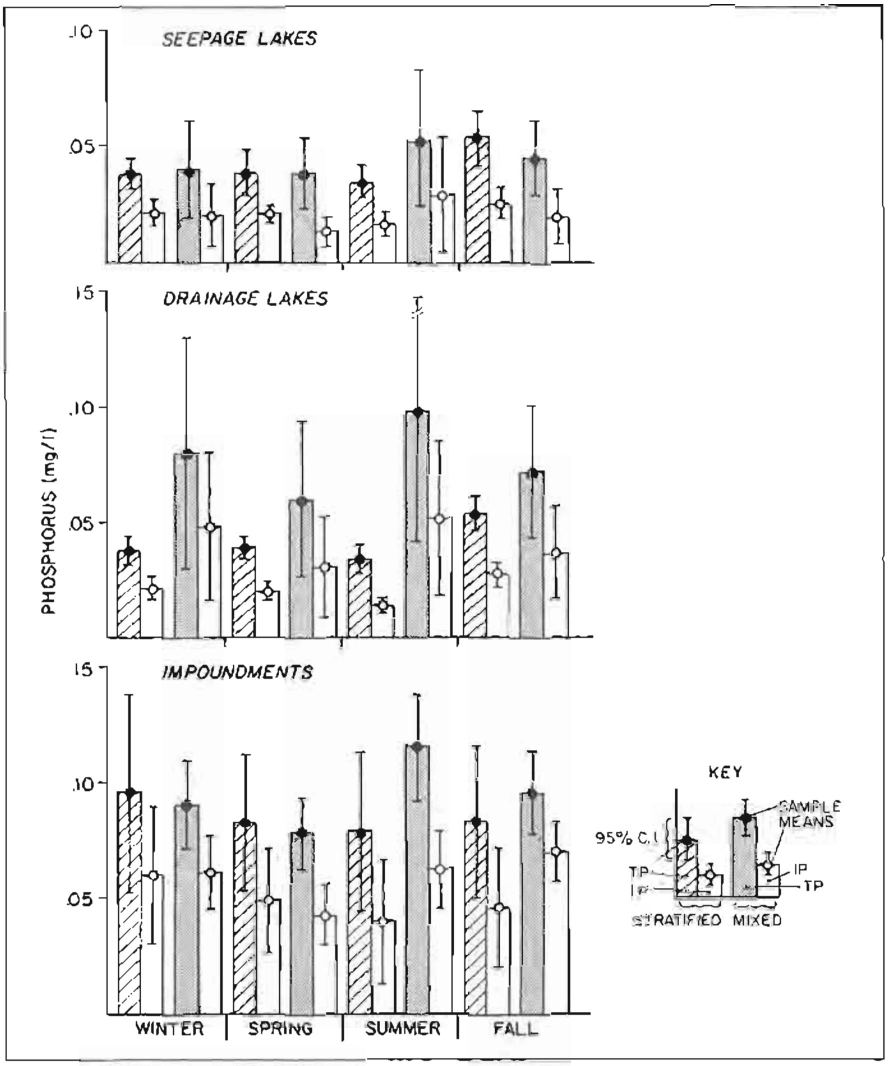


FIGURE 62. *Seasonal means for inorganic and total phosphorus in different lake types (quarterly data set).*

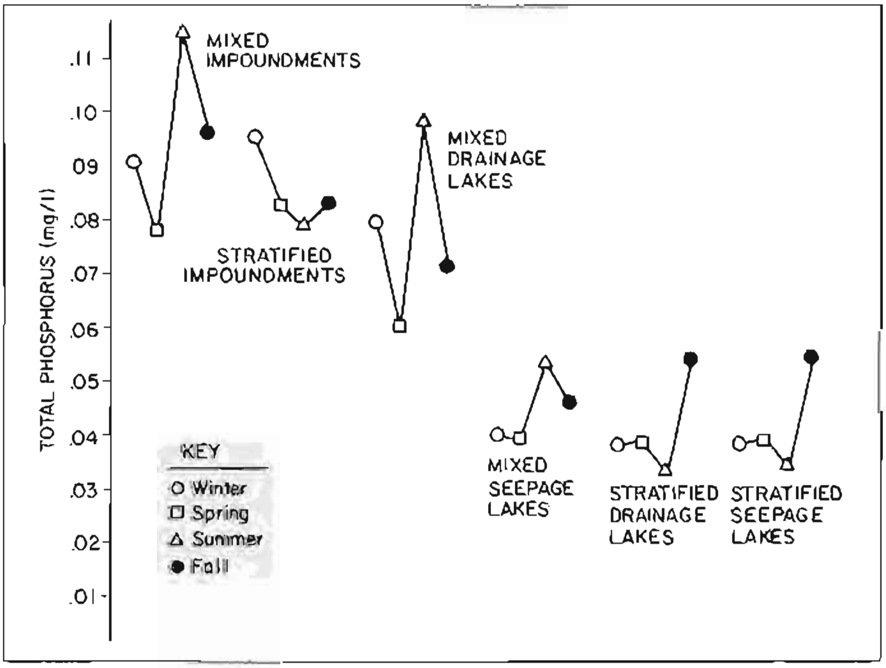


FIGURE 63. *Seasonal total phosphorus means for six lake types (quarterly data set).*

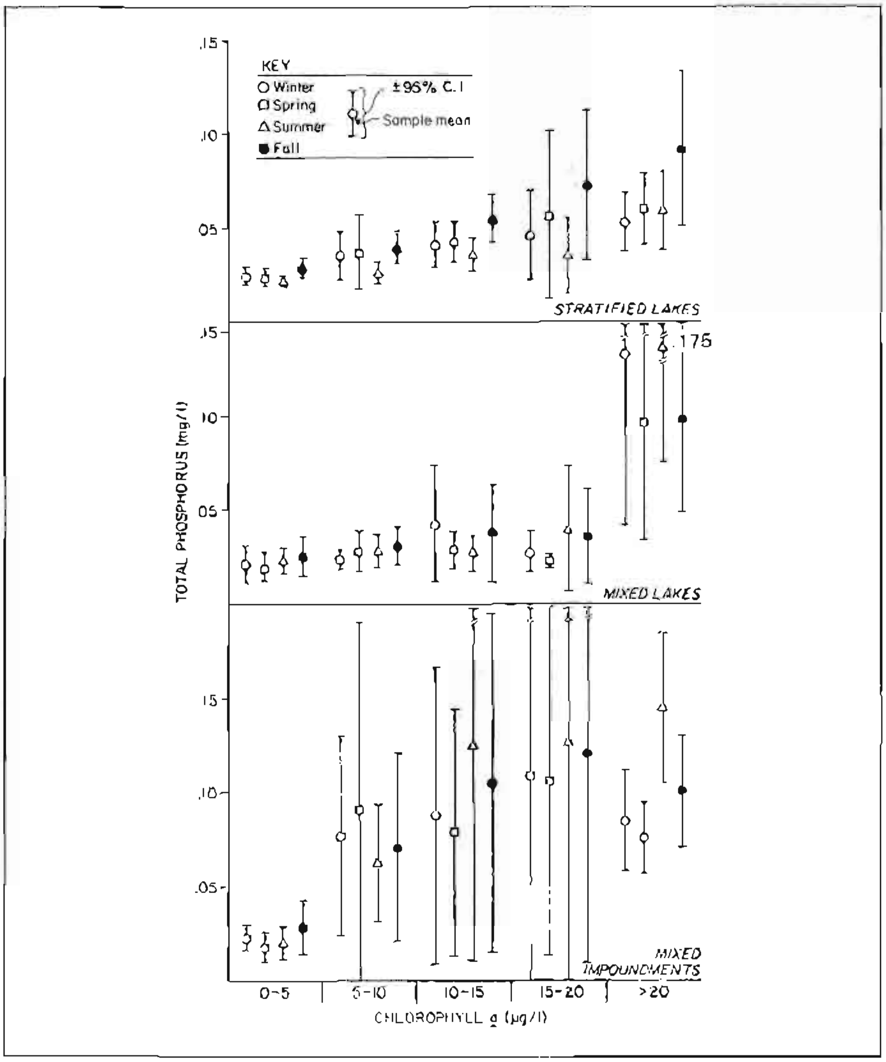


FIGURE 64. *Seasonal changes in total phosphorus as related to lake type and chlorophyll* a *concentrations (quarterly data set).*

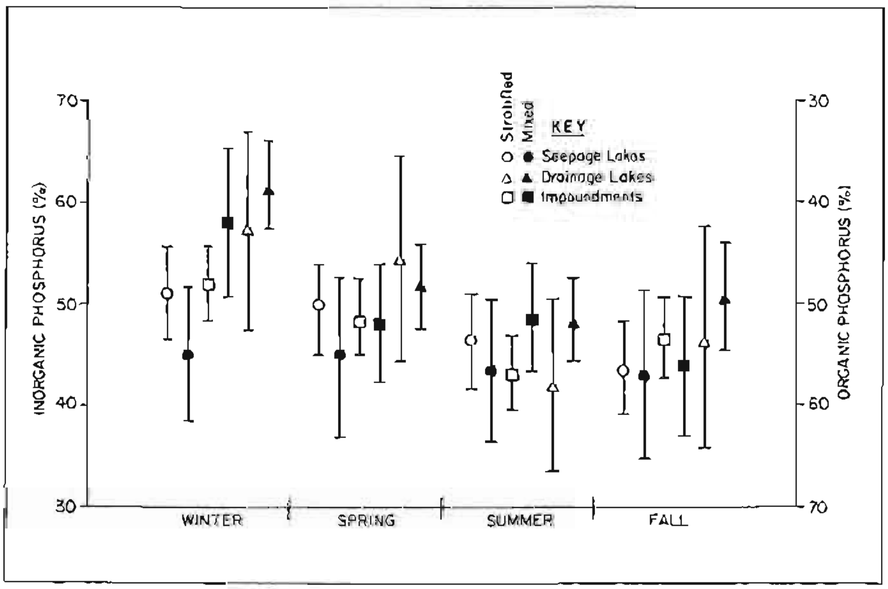


FIGURE 65. *Seasonal changes in inorganic and organic phosphorus means ($\pm$95% C.I.) for different lake types (quarterly data set).*

percentage of organic phosphorus than natural lakes; this may be due either to higher turbidities inhibiting the biological conversion of inorganic phosphorus into organic cell matter, or high flow-through rates limiting the time available for this conversion to take place. In effect, the impoundments more closely resemble the streams which are their source.

Seasonal phosphorus means are related to summer chlorophyll *a* levels and lake type (Fig. 64). As shown previously, low phosphorus levels correspond to low chlorophyll *a* levels. Also, the decrease in summer total phosphorus in stratified lakes, referred to earlier, is evident in each chlorophyll *a* class. The fall total phosphorus sample means are in line with the trend shown by the winter and spring levels for stratified lakes. The opposite appears to be true for mixed lakes and impoundments. Mixed lakes and impoundments show much more variability in their total phosphorus-chlorophyll *a* relationship. Impoundments appear to require higher levels of phosphorus to produce given amounts of chlorophyll *a* than stratified lakes (Fig. 64). At least this appeared to be true within the 5-20 µg/l chlorophyll *a* range. While this might be expected as a result of such factors as light inhibition (self-shading or high turbidities) or hydraulic washout prevalent in high flow-through impoundments, it is contradictory to earlier discussions concerning the relationship of chlorophyll *a* and phosphorus (pages 61-62) in impoundments, mixed lakes and drainage lakes, which suggested that the opposite was true. No plausible explanation comes to mind; possibly the fact that this "seasonal" discussion is based on a different data set (quarterly data set) is of significance.

The same seasonal trends found statewide in stratified and mixed lakes and impoundments were also found on a regional basis. Regional comparisons are not considered to be significant because they are believed to reflect gross differences in the physical and chemical nature of the lakes in the different regions rather than trends resulting from climatic differences. Generally, the lakes with the highest total phosphorus levels (more eutrophic) showed the greatest variations between seasons.

Examination of the effect of various morphological characteristics on the seasonal changes in total phosphorus was restricted by the large variance in the data. However, certain generalizations can be made. While there was no significant difference between the mean depth of impoundments experiencing either an increase or decrease in total phosphorus from spring to summer, natural lakes showing an increase had significantly lower mean depths than lakes showing a decrease. Juday and Birge (1931) reported finding a similar seasonal relationship of mean depth and total phosphorus. Our data also showed that total phosphorus varies considerably less from spring to summer in deep lakes than in shallow lakes, which is not a surprise considering the fact that the deeper lakes generally have lower total phosphorus values overall.

The ratio of mean depth to maximum depth (a ratio greater than 0.33 indicates a concave lake basin form vs a convex basin form for lakes with a ratio less than this value) appeared to be slightly higher in lakes showing an increase in total phosphorus from spring to summer, but the difference was not statistically significant. This characteristic of basin morphometry and its possible subtle effect on the change in total phosphorus may be related to its impact on stratification. A plot of lakes coded as to stratification using the basin form index vs lake area indicates that between lakes of nearly equal areas and maximum depths, the lake with a convex basin will stratify. Thermal stratification, therefore, appears to be the deciding factor affecting total phosphorus concentrations in some lakes.

Slope also appears to be important. Lakes showing decreases in total phosphorus from spring to summer had steeper slopes (associated with greater maximum depths) than lakes experiencing an increase. There was evidence that stratified lakes with large percentages of bottom area exposed to the epilimnion experience increases in total phosphorus from spring to summer. The reverse situation was also observed; total phosphorus generally decreased from spring to summer in stratified lakes that have small percentages of bottom area exposed to epilimnetic mixing and resuspension.

The changes in inorganic and total phosphorus for lakes with various drainage basin to lake area ratios are not consistent (Table 34). While means for lakes with smaller ratios are relatively low, the percent inorganic-organic phosphorus fraction remains fairly constant from season to season and with differing ratios. However, lakes with very large DB:LA ratios generally have slightly higher percentages of inorganic phosphorus throughout the year.

Significant correlations were found (and linear regression equations computed) for the relationships between the different seasonal phosphorus means for the various lake types. However, as Reckhow (1979) has reported, a few extreme values may be given more weight than a large number of clustered values in the determination of correlation coefficients and regression equations where data are not normally distributed. This proved to be a considerable problem in the evaluation of seasonal comparisons since the quarterly data set included a small number of extremely high values. Elimination of the lakes with extreme values and recomputation of the correlations resulted in a significant reduction in the correlation coefficients and had a significant negative impact upon the regression analysis equations. Plots of the relationships showed considerable scatter, indicating little in the way of usefulness other than to support general well-known associations.

TABLE 34. *Seasonal means for nitrogen and phosphorus in lakes with different drainage basin:lake area ratios.*

Season		DB:LA < 10	DB:LA 10-50	DB:LA 50-100	DB:LA 100-1000	DB:LA > 1000	All
	Organic N	0.48	0.59	0.46	0.48	0.48	0.52
	Total N	0.85	1.17	1.15	1.68	1.41	1.14
Winter	Inorganic P	0.019	0.033	0.030	0.073	0.058	0.035
	Total P	0.036	0.059	0.049	0.100	0.097	0.057
	Organic N	0.48	0.58	0.54	0.64	0.58	0.55
	Total N	0.74	1.05	1.11	1.45	1.38	1.02
Spring	Inorganic P	0.016	0.024	0.024	0.049	0.049	0.026
	Total P	0.033	0.050	0.047	0.084	0.083	0.051
	Organic N	0.54	0.72	0.69	0.79	0.60	0.65
	Total N	0.74	1.08	1.01	1.27	1.18	0.97
Summer	Inorganic P	0.019	0.028	0.022	0.058	0.061	0.031
	Total P	0.039	0.059	0.045	0.108	0.109	0.061
	Organic N	0.52	0.65	0.63	0.58	0.56	0.59
	Total N	0.78	0.99	1.02	1.23	1.38	0.98
Fall	Inorganic P	0.022	0.031	0.027	0.050	0.078	0.032
	Total P	0.046	0.067	0.057	0.092	0.124	0.065
	N	213-264	110-126	29-32	67-80	19-21	438-523

Nitrogen dynamics

Nitrogen appears to be a relatively static parameter as compared to phosphorus (Table 33). While total nitrogen concentrations (means) are significantly different between data sets, coefficients of variation for nitrogen are fairly uniform from season to season and are considerably lower than those for phosphorus. This relative stability may be of considerable importance in the development of lake sampling programs.

The seasonal means of total and organic nitrogen concentrations of Wisconsin lakes vary according to lake type classifications (Fig. 66). Stratified seepage and drainage lakes have the lowest total nitrogen and organic nitrogen means. As the phosphorus analysis showed, overlap in confidence intervals limit the ability to make definitive statements, but trends are suggested by the sample means. Mixed seepage and drainage lakes show a decrease in total nitrogen from winter to spring, followed by a slight increase into summer and a decline in the fall. Mixed impoundments show a continuous decrease from winter to the following fall. This corresponds to the findings of stream studies which show a continuous decrease in nitrogen throughout the year from high winter levels (Lathrop and Johnson 1979, Mason unpubl. data). Stratified drainage lakes show fairly uniform total nitrogen levels throughout the year with small increases of organic nitrogen in the summer and fall. Both the mixed and stratified seepage lakes demonstrate small increases in organic nitrogen to summer maxima, followed by a decline in the fall. The net increases in organic nitrogen throughout the growing season may reflect temperature-dependent biological conversion of inorganic nitrogen into organic forms.

Some significant differences existed in the percent of organic-inorganic nitrogen between the various lake groups (Fig. 67). A significant difference was found between mixed seepage lakes and stratified seepage lakes, but only during the fall. In both instances, the mixed lakes had a higher percentage of organic nitrogen. This appears to be the result of an increase in the inorganic nitrogen fraction in the stratified lakes, possibly due to the "recent" mixing of the rich hypolimnetic waters with the epilimnion. As Figure 66 shows, there is a very little change in organic nitrogen from summer to fall in stratified lakes, while total nitrogen increases. Mixed lakes show a decrease in total nitrogen and organic nitrogen while the ratio of inorganic to organic

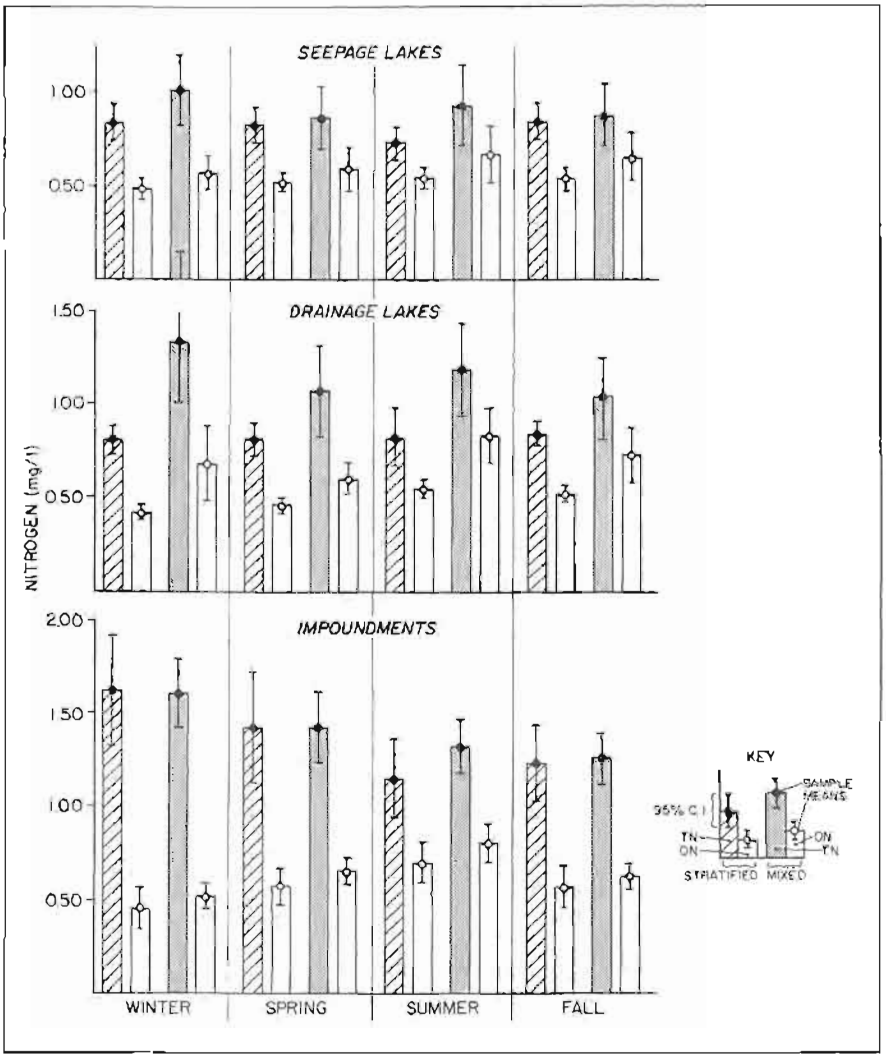


FIGURE 66. *Seasonal means of total and organic nitrogen for different lake types.*

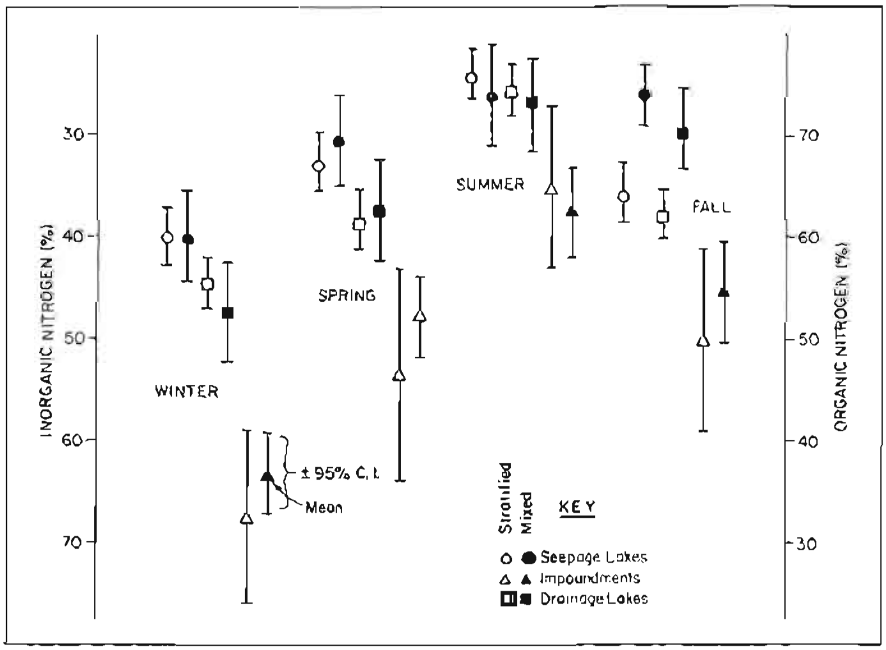


FIGURE 67. *Seasonal changes in inorganic and organic nitrogen levels in different lake types (quarterly data set).*

nitrogen remains essentially the same. While not significant at the 95% confidence level, drainage lakes (mixed or stratified) as a group consistently had a lower percentage of organic nitrogen than seepage lakes. Likewise, mixed and stratified impoundments had a significantly lower percentage of organic nitrogen (conversely higher inorganic nitrogen percentages) than either seepage or drainage lakes in all seasons, with the exception of stratified impoundments during the summer. These differences again reflect both the character of the water source and the retention time of the particular water body. Absolute values of total nitrogen and inorganic nitrogen were generally higher in impoundments and drainage lakes than in seepage lakes. These high absolute values and percentages of inorganic nitrogen in impoundments and drainage lakes indicate large influxes of nitrogen from the watershed and relatively small or poor conversion of inorganic to organic nitrogen within these water bodies.

Omernik (1977) found in a study of nutrient export from different types of watersheds that the inorganic nitrogen fraction of the total nitrogen concentration decreased from heavily agricultural watersheds to more highly forested watersheds. Our Wisconsin lake data (Table 34) support his findings; percent inorganic nitrogen decreased with a decrease in drainage basin:lake area ratio, and lakes with smaller DB:LA ratios generally were located in more heavily forested watersheds. Omernik (1977) reported a mean of 38% inorganic nitrogen concentration for streams draining watersheds in the Eastern region of the United States, which corresponds well with the 27-49% inorganic nitrogen component found in lakes of differing DB:LA in Wisconsin.

Wisconsin lakes with relatively large watersheds (as indicated by large DB:LA ratios) have consistently higher inorganic nitrogen and total nitrogen content than lakes with small watersheds (Table 34). The seasonal means for organic nitrogen concentrations range from 0.48 mg/l to 0.79 mg/l, with peak values occurring during the summer season. Inorganic nitrogen (total nitrogen less organic nitrogen) shows much greater variation; largest mean values (and percentages of total nitrogen) are found in lakes with larger DB:LA ratios. Inorganic nitrogen decreases from winter to summer (regardless of DB:LA ratio) by as much as 60% and rises again in the fall. The percent inorganic-organic fraction varies in a similar manner, reflecting the conversion of inorganic to organic matter.

All lake types followed the same general pattern, with lowest percent-

TABLE 35. *Number of lakes of various classifications experiencing either an increase or decrease in total nitrogen from one season to the next.*

	Impoundments								Seepage Lakes								Drainage Lakes							
	Mixed				Stratified				Mixed				Stratified				Mixed				Stratified			
No. lakes showing change in Total N																								
Increase	25	34	33	51	6	8	12	13	13	25	14	25	44	32	59	49	16	31	20	34	61	46	72	48
No change	2	0	0	1	1	0	0	0	0	0	0	0	4	1	1	1	0	1	1	0	3	5	2	1
Decrease	46	39	40	21	12	11	7	6	26	14	25	14	47	62	35	45	34	18	29	16	50	63	40	65
Season to Season	W-Sp	Sp-S	S-F	F-W	W-Sp	Sp-S	S-F	F-W	W-Sp	Sp-S	S-F	F-W	W-Sp	Sp-S	S-F	F-W	W-Sp	Sp-S	S-F	F-W	W-Sp	Sp-S	S-F	F-W
Total No. Lakes	73				19				39				95				50				114			

ages of organic nitrogen occurring during the winter and peaks or maxima occurring during the summer, followed by a decline in the fall.

No significant correlations were found between seasonal levels of total nitrogen and either lake size or volume, but there were fairly good negative correlations between total nitrogen and mean or maximum depths. This implies that given two lakes of equal volume, the lake with the greater mean depth is likely to have a lower total nitrogen concentration (epilimnion). Also, in the case of two lakes having equal surface acreages, the lake with the greater mean depth (or maximum depth, which is closely associated with mean depth) probably would have the lower total nitrogen level. The reason why depth appears to correlate better with total nitrogen levels than either lake volume or area is uncertain; undoubtedly watershed size, gradient, land use and percent runoff are important factors influencing the evaluation.

Although changes in total nitrogen from season to season are dependent upon lake type (Fig. 68), the net change in total nitrogen is of little significance since these values are related to the number of lakes in each lake type increasing or decreasing in total nitrogen from one season to the next (Table 35). Some patterns appear when comparing mixed seepage and drainage lakes with stratified seepage and drainage lakes. The net change in total nitrogen from season to season is clearly dependent upon the number of lakes increasing or decreasing in total nitrogen. More mixed lakes show alternating changes, e.g., decrease from winter to spring and increase from spring to summer, than stratified lakes where the change from winter to spring is about evenly divided, but the majority show a decrease from spring to summer.

Mean depth apparently was not a major factor influencing seasonal nitrogen changes in lakes or impoundments; the mean depths of lakes showing increases in total nitrogen from spring to summer are not significantly different from those lakes showing decreases

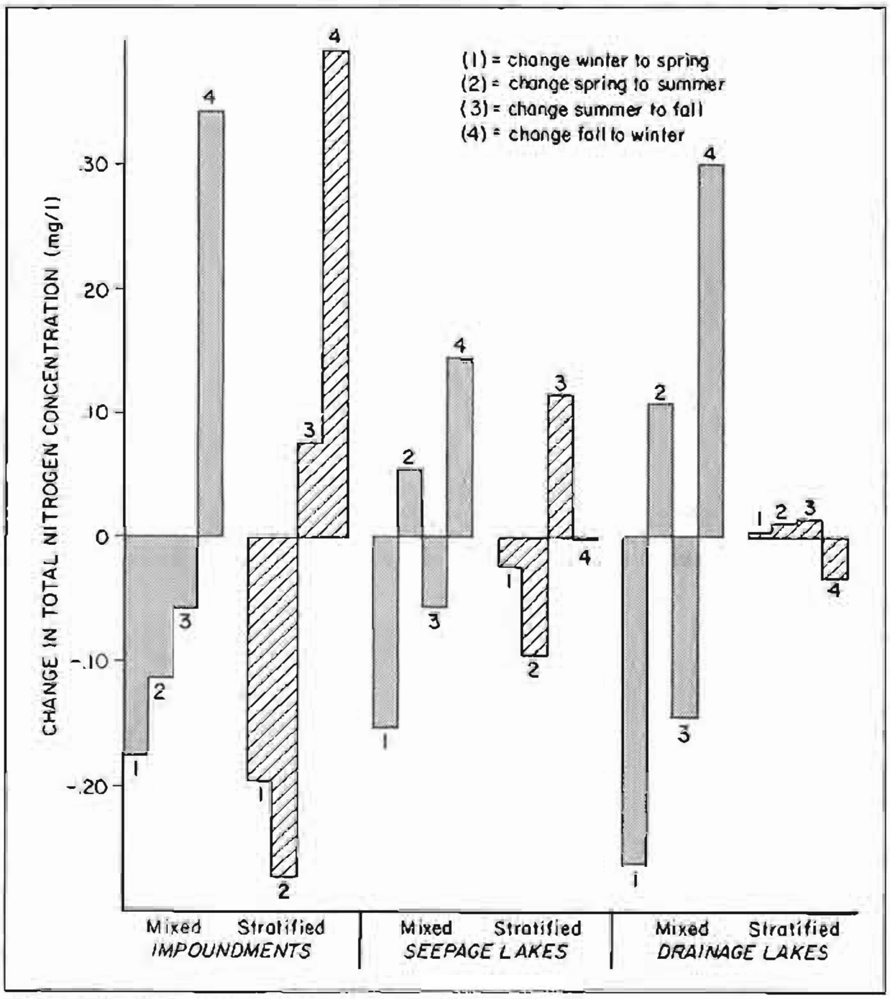


FIGURE 68. *Net seasonal change in total nitrogen in different lake types (quarterly data set).*

(within similar lake types, Fig. 69). This was found to hold true for the other seasonal changes as well. Thus, the direction of changes in total nitrogen appear to be independent of mean depth while the mean concentrations (absolute) of total nitrogen are inversely related to mean depth.

Total Nitrogen:Total Phosphorus Ratios

Concentrations of phosphorus and nitrogen are generally highly associated with one another and are also associated in differing ways with all other measured parameters (negatively correlated with water clarity and depth, see Append. B). These associations show a great deal of variability between data sets and lake types. As demonstrated in the previous sections, seasonal changes in nutrient concentrations are greatly dependent upon lake type, physical features of the lakes, and characteristics of the lake watershed. Nutrient dynamics are of great importance in lake management and considerable effort has gone into establishing

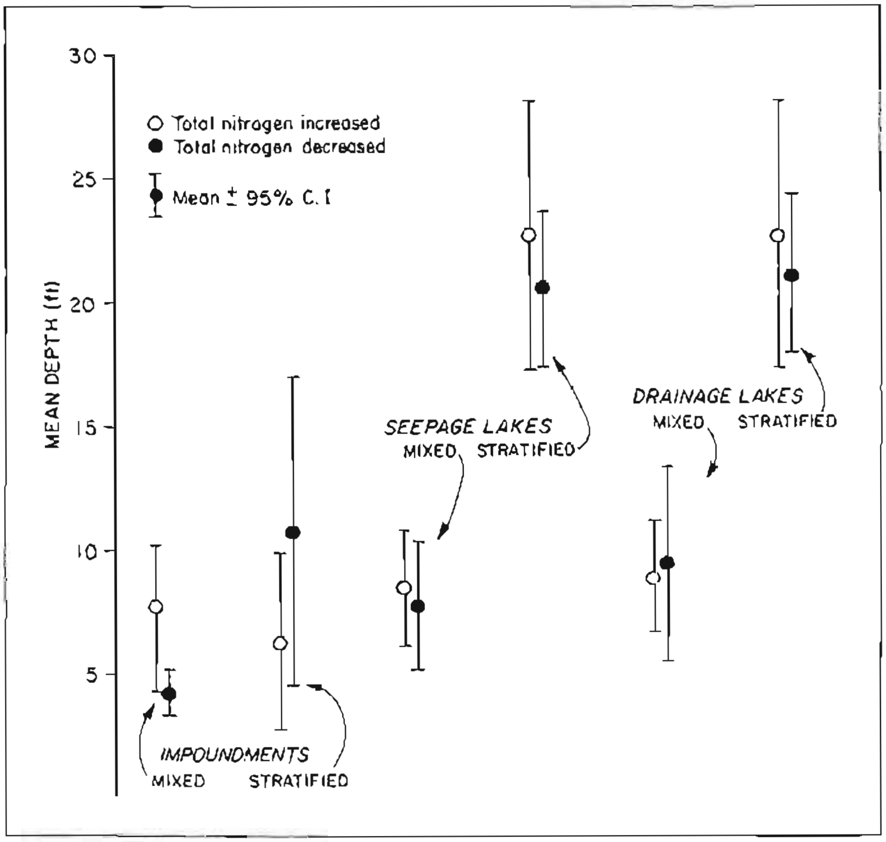


FIGURE 69. *Mean depths of different lake types separated on the basis of an increase or decrease of total nitrogen from spring to summer (quarterly data set).*

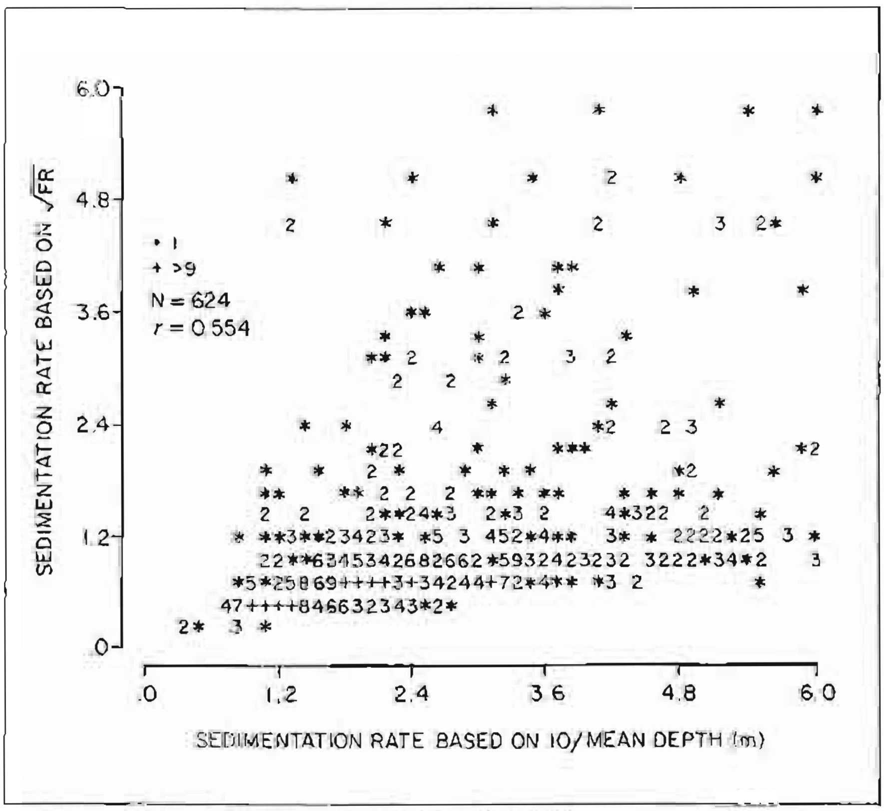


FIGURE 70. *Relationship to specific sedimentation rate (6) estimates based on $\sqrt{FR}$ and 10/mean depth in meters (161 lakes not shown; total data set).*

a cause-effect relationship between nutrient concentrations and water quality (previously discussed in other sections).

Because of the influence of plant nutrients (in particular phosphorus) on lake water quality, various in-lake concentrations and watershed external loading limitations have been suggested as "standards" or necessary levels for maintaining good water quality conditions. The ratio of nitrogen to phosphorus has also been stressed as having significant impact upon lake water quality (see discussion p. 63-64).

Further analysis of the N:P ratios and their relationship to other water quality characteristics seems to indicate that most Wisconsin lakes are phosphorus limited and nitrogen limitation is found in only a few lakes (Table 36). Generally, lakes with high N:P ratios had good water quality, 65% of low N:P lakes had Secchi disc readings less than 2 m opposed to only 43% of the high N:P ratio (>40) lakes. Also, chlorophyll *a* was lower in lakes with high N:P ratios than in low N:P lakes; only 16% of the lakes with N:P ratios greater than 40 had chlorophyll *a* greater than 15 µg/l as compared to 27% of the lakes with lower N:P ratios.

Mean total phosphorus for high N:P ratio lakes was only 0.015 mg/l (Append. A) with 80% of these lakes having values equal to or less than 0.015 mg/l. Low N:P lakes were generally high in total phosphorus, which is consistent with the findings of Lambou et al. (1976). Means for total nitrogen and organic nitrogen were slightly higher in lakes with N:P greater than 40 than in lakes with N:P of 15-40, but these means overlapped those from lakes with N:P less than 15 (Append. A).

Color was slightly higher in high N:P lakes, but this may have been due to its association with nitrogen. Alkalinity, pH and magnesium means showed similar relationships with N:P ratio. No low N:P lake had a pH less than 6.0. Of particular interest may be the gradual decrease in magnesium concentration with increasing N:P ratio, followed by a slight increase when N:P exceeded 40 (Append. A). This may be associated with the negative correlation of total phosphorus and pH, and/or magnesium and total nitrogen in the dolomite area of southeastern Wisconsin. Our data showed the mean N:P ratio for the Southeast Region lakes was 41:1, but Gerloff and Skoog (1957) noted the presence of a yellowish-green color in some lakes in southeastern Wisconsin which were apparently nitrogen limited. A similar color was observed in some of the lakes in our sample of the Southeast Region which had low N:P ratios.

Applicability of Phosphorus Models to Wisconsin Lakes

Phosphorus is generally recognized as the most important nutrient contributing to the eutrophication process in lakes; thus, understanding its role and controlling its impact are matters of great interest. Attempts to model phosphorus dynamics in lake ecosystems throughout the world have gained considerable attention. Phosphorus models have been used extensively in lake management efforts as predictive tools for determining sources and consequences of phosphorus loading to lakes and evaluating proposed lake protection or restoration projects. Because many different models have been developed based on various data sets and influencing factors, evaluations of the precision and "best-fit" regressions of these models have been made (Uttormark and Hutchins 1978, 1980). Application of these models to Wisconsin lakes has not been extensively tested, primarily because data are sparse for key parameters. Paramount among these deficiencies is the lack of accurate loading data, which are available for only a few Wisconsin lakes. Therefore, loading rates for nearly all lakes in our data base have to be estimated in order to utilize the phosphorus models. Flushing-rate data are also important in phosphorus modeling, and are mostly unavailable for Wisconsin lakes and must be estimated in model application.

In phosphorus modeling, estimates of total in-lake phosphorus concentrations are derived by two basic means: (1) computation based on knowledge of the specific areal phosphorus loading (L), mean depth, and flushing rate (FR = 1/RT), or (2) computation based on average inflow phosphorus concentrations and flushing rate. In the case where L is measured, an alternative means of estimating the total in-lake phosphorus concentration using a phosphorus retention coefficient (R) has been has been developed by Dillon and Rigler (1974). The phosphorus retention coefficient, which is the percentage of the incoming phosphorus trapped within a lake, is in turn derived from the flushing rate and the specific sedimentation rate. The specific sedimentation rate may be estimated based on (1) mean depth, or (2) the lake's flushing rate. In applying phosphorus modeling to our lake data, specific sedimentation rates for the lakes were calculated by both methods and, although they were strongly correlated ($r = 0.554$), individual lakes showed considerable variation (Fig. 70). Greatest disparity between methods was noted in natural lakes while the sedimentation rates for impoundments were quite similar (Table 37). Phosphorus retention coefficients based on specific sedimentation rates computed from flushing rates ($\sqrt{FR}$) showed a higher degree of correlation with observed phosphorus levels ($r = -0.331$ spring; $r = -0.299$ summer) than correlations based on specific sedimentation rates computed from mean depth (10/mean depth in meters) ($r = -0.268$ spring; $r = -0.231$ summer). This finding may be of particular significance in modeling efforts inasmuch as the former estimates of retention coefficients were based on "estimates" of flushing rates while the latter estimates were based on "known" mean depth values.

Further evaluation of the various phosphorus models as they pertain to Wisconsin lakes is severely restricted as mentioned before, due to lack of adequate loading data for most lakes. Although the models are based on mass balances and long-term averages of in-lake data, back-calculation of estimated phosphorus inflow concentrations based on the relationship:

$$P_{inflow} = P_{in\text{-}lake} \div \frac{FR}{\sigma + FR}$$

Sedimentation Rate: case 1 $\sigma = \sqrt{FR}$; case 2 $\sigma = 10 \div$ mean depth in meters

(where $P_{in\text{-}lake}$ = mean annual total phosphorus concentrations and FR = the reciprocal of the retention time) provide some indication of the applicability of models to Wisconsin lakes (Table 38). Based on the model, mean annual in-lake phosphorus concentrations should always be less than mean annual inflow phosphorus concentrations.* Because mean annual phosphorus concentrations were not determined, and spring and

TABLE 36. *Relationship between lakes with various total nitrogen:total phosphorus ratios and other water quality parameters.*

		N:P Ratios							
		<10		10-15		15-40		>40	
		No. Lakes	%	No. Lakes	%	No. Lakes	%	No. Lakes	%
Chlorophyll a (µg/l)	0-5	6	23	7	23	78	26	96	33
	5-10	5	19	9	30	111	37	107	37
	10-15	3	12	3	10	37	12	37	13
	15-25	3	12	2	7	38	13	27	9
	>25	9	35	9	30	37	12	20	7
Secchi disc (m)	0-1	6	23	11	41	57	20	32	12
	1-2	11	42	3	11	82	29	80	31
	2-3	2	8	6	22	73	25	78	30
	3-4	4	15	4	15	42	15	39	15
	4-5	1	4	2	7	19	7	21	8
	5-6	2	8	0	0	9	3	4	2
	>6	0	0	1	4	5	2	6	2
pH (units)	<5	0	0	0	0	4	1	3	1
	5.0-5.9	0	0	6	19	24	8	23	8
	6.0-6.9	4	15	1	3	94	30	88	30
	>7	23	85	25	78	189	61	180	61
Total alkalinity (mg/l)	0-15	4	15	6	19	111	36	101	34
	15-30	1	4	4	12	48	15	53	18
	30-90	14	52	15	47	96	31	67	23
	>90	8	30	7	22	56	18	72	26
Total phosphorus (µg/l)	<5	0	0	0	0	11	4	93	32
	5-15	2	7	2	6	92	29	136	46
	15-25	2	7	4	12	88	28	45	15
	25-35	1	4	5	16	51	16	14	5
	>35	22	81	21	66	70	22	5	2

TABLE 37. *Median sedimentation rates for impoundments and natural lakes in Wisconsin.*

Sedimentation Rate Based on:	Impoundments	Natural Lakes
FR	5.8	1.1
10/mean depth (m)	5.4	2.4

TABLE 38. *Estimated phosphorus inflow concentrations to Wisconsin lakes on a regional basis.*

Region	Estimated Phosphorus Inflow Concentrations (µg/l)			Observed In-lake Phosphorus Concentrations in µg/l		
	Omernik*	Est#1	Est#2	Impds.	Lakes	All
Northeast	12	63	134	59	24	27
Northwest	17	66	127	53	32	35
Central	40	60	114	51	24	36
Southeast	150	139	282	173	65	84
Southwest	60	126	143	101	58	98

*Based on Omernik (1977)
Est #1 where $\sigma = \sqrt{FR}$
Est #2 where $\sigma = 10 \div$ mean depth (m).

fall turnover data were not available for many sampled lakes, summer total phosphorus concentrations were used as a rough approximation of the mean annual in-lake phosphorus concentrations. As previously discussed, these summer total phosphorus values are probably higher than annual mean values in impoundments and mixed lakes and may be similar or slightly lower than annual means in stratified lakes.

The impact of the composition of lake types (e.g., number of impoundments vs stratified lakes) upon the measured total phosphorus values and the resultant "predicted" phosphorus loading concentrations is readily apparent in the Southwest Region. (Table 38). Mean measured in-lake total phosphorus (98 µg/l) and estimated inflowing phosphorus (126 µg/l) greatly exceed Omernik's (1977) observed instream concentrations (60 µg/l) (almost all of the lakes in this region are impoundments where summer total phosphorus values greatly exceed spring values which may be a closer approximation of annual means).

Data for the Southeast and Central regions appear reasonable, while measured in-lake phosphorus concentrations for the Northeast and Northwest regions appear to be higher than would be expected based on Omernik's stream phosphorus concentrations. Because a high percentage of the lakes in the northern regions are natural, deep, stratified lakes, it seems logical that measured summer epilimnetic total phosphorus values would most likely be lower than annual means, and thus lead to underestimation in the back-calculation of annual phosphorus loading from watersheds. If this is the case, then the estimated phosphorus loadings based on Omernik's data for the regions are probably lower than they should be . However, it is more likely that a few lakes with high in-lake phosphorus concentrations skewed the mean for the regions, which results in the in-lake concentrations exceeding the actual average inflow concentrations, and in overestimation of the inflow phosphorus concentrations.

Estimated inflow concentrations based on sedimentation rate coefficients (σ) calculated by the $\sqrt{FR}$ more closely relate to Omernik's instream concentrations than do similar estimates based on sedimentation coefficients calculated by 10 ÷ mean depth (Table 38).

Based on the in-lake phosphorus data collected during our sampling program and our estimates of hydraulic loading and retention time for sampled lakes, it is not possible to reliably predict watershed phosphorus contributions to lakes. Conversely, it also appears that in-lake phosphorus concentrations (summer) cannot in most cases be accurately predicted by using regional stream concentrations. Once again, the complexity of Wisconsin lake types and characteristics make them extremely difficult to categorize, and it appears that phosphorus models will fit reasonably well only for individual lakes on which adequate data are available.

INTERRELATIONSHIPS BETWEEN LAKE CHARACTERISTICS

OVERVIEW

General relationships between various water quality parameters have been reported upon extensively in limnology texts (Hutchinson 1975, Wetzel 1975). However, because the large number of lakes sampled in this study spanned a wide range of lake types and conditions, the opportunity exists for a more detailed analysis of these relationships.

It is important to stress that these comparisons are not all cause-effect relationships, but perhaps more correctly represent associations. Even in cases where significant correlation coefficients are found between theoretically valid relationships, other unidentifiable factors may be responsible for the associations.

Reckhow (1979) explicitly describes the weaknesses involved in the use of correlation coefficients and linear regressions: primarily these deal with the assumption that the data are indeed *linear* and that the data are *normally* distributed. As will be shown later, these assumptions are not always valid, and even the transformation of the data to compensate for these problems does not always create a totally unbiased relationship.

Relationships between different water quality characteristics are provided in Appendix B. The computer program used to create the correlation matrixes from which these figures were drawn did not print the number of matching pairs of data points used in computing the correlation coefficients. In order to make comparisons of significance between matrixes (data sets), it was necessary to compute the number of degrees of freedom based on the minimum number of data points for any particular parameter within a given data set. Therefore, the levels of significance presented in Appendix B are somewhat conservatively labeled, but are quite adequate for making comparisons and generalizations between and about the various subsets since the data sets are quite large. The following discussion is based upon the data generated in Appendix B.

Since interrelationships between parameters graphically displayed in Appendix B are readily observable, only major highlights will be specifically mentioned. The following format lists each major characteristic (e.g., Area) and, on the left margin, other characteristics with which it is closely associated, either positively or negatively in almost all of the subsets of lakes analyzed (e.g., mean depth). The characteristics that are indented (e.g., maximum depth) apply only to the restricted groups of lakes indicated in the parentheses (e.g., seepage lakes and impoundments). Some associations important on a regional basis will also be mentioned.

AREA

+ Mean depth
  + Maximum depth (seepage lakes and impoundments)
  + Alkalinity (low alkalinity lakes)
  + Shoreline development factor (seepage lakes, impoundments and low chlorophyll *a* lakes)

Overall, area showed generally poor correlation with other lake characteristics. In the Northeast, there were positive relationships between area and maximum depth, and area and inorganic and total phosphorus. There was a negative correlation between area and Secchi disc, and positive correlations with inorganic and total phosphorus in the Southeast Region. The Central Region had positive correlations between area and chlorophyll *a* and total phosphorus. The Southwest Region showed no significant relationship between area and other lake characteristics.

MEAN DEPTH

+ Area
+ Maximum depth
+ Water clarity
− Turbidity
− Nutrients
  − Color (drainage lakes)
  − Chlorides, calcium, magnesium (impoundments)
  − Chlorophyll *a* (seepage lakes and low alkalinity lakes)

The significance of the relationship between mean depth and maximum depth was discussed previously under General Characteristics — Physical Features. The relatively strong negative correlation of mean depth with calcium, magnesium, alkalinity and chlorides in impoundments shows the influence of lake volume and flushing rate on these associations.

The strongest positive correlation of mean depth and water clarity was in the Central Region. Negative correlations between mean depth and color, chlorophyll *a*, turbidity, and organic and total nitrogen were evident in the Northeast Region. Phosphorus showed little association with mean depth in the Northeast Region, but a weak negative correlation between mean depth and nitrogen and total phosphorus was observed in the Northwest Region.

MAXIMUM DEPTH

+ Mean depth
+ Water clarity
− Nitrogen
  − Chlorides, inorganic phosphorus (impoundments)
  − Turbidity (seepage lakes, drainage lakes and low alkalinity lakes)
  − Total phosphorus (drainage lakes)
  − Color (drainage lakes)
  − Chlorophyll *a* (low alkalinity lakes and low chlorophyll *a* lakes)

The association of maximum depth with nitrogen is generally more pronounced than its association with phosphorus. This may be related to the higher coefficients of variation in the phosphorus concentrations (See Table 33). A rather unusual relationship exists between maximum depth, mean depth and area. As demonstrated earlier, maximum depth and mean depth were often highly related, as were area and mean depth. Yet, where these relationships were the strongest, the relationships between maximum depth and area were at best only weak. The inverse of this situation was also true; the maximum depth-area relationship was strongest where the mean depth-area relationship was weakest. The reasons for, and the significance of, this apparent anomaly are unknown. Both natural lakes and drainage lakes are similar in this respect. The significance of maximum depth as it relates to stratification and the channeling of nutrients in lake systems was discussed earlier (General Characterstics — Lake Morphometry).

Water clarity was positively correlated with maximum depth in all regions except the Central Region. Chlorophyll *a*, turbidity and nitrogen were all inversely related to maximum depth in the Northeast Region. Nitrogen and phosphorus were weakly related to mean depth in the Northwest and Southeast regions, but such was not the case in the Central and Southwest regions. A positive correlation of pH with mean depth was also found in the latter two regions.

SHORELINE DEVELOPMENT FACTOR

− No overall strong correlations
  + Area (seasonal data, impoundments, seepage lakes and low chlorophyll *a* lakes)
  + Mean depth (impoundments)
  + Maximum depth

- Magnesium, alkalinity (seasonal data and impoundments)

The significance of the shoreline development factor (the ratio of the shoreline perimeter divided by the circumference of a circle with the same area as the lake) and its relationship to other water quality characteristics does not appear to be great. Other factors (mean depth, watershed size, etc.) apparently outweigh the impact of this parameter.

COLOR

- − Water clarity
  - − Mean depth (drainage lakes)
  - − Maximum depth (drainage lakes)
  - − pH (drainage lakes)
  - + Chlorophyll *a*, chlorides, calcium (low chlorophyll *a* lakes and low alkalinity lakes)

Strong correlations between color and water clarity were observed in all subsets of data (see also discussions by Anthony and Hayes 1964). A weak association with nitrogen is evident in most subsets (see also Append. A).

Some regional differences concerning color were noted. Secchi disc and color were strongly related in the Northeast and Northwest regions, but other factors were apparently more important or overriding in the other regions. Color was also strongly associated with chlorophyll *a* in the Northeast Region and with organic and total nitrogen in both the Northeast and Northwest regions. In the Southeast Region, color was strongly associated with calcium, magnesium and alkalinity, while in the Southwest Region the opposite was true. In both regions, color was inversely related to pH.

WATER CLARITY (SECCHI DISC DEPTH)

- + Mean depth
- + Maximum depth
- − Color
- − Chlorophyll *a*
- − Turbidity
- − Nutrients
  - + pH (impoundments)
  - − chlorides (impoundments)
  - − chlorides, calcium, magnesium, pH and alkalinity (low color lakes)

The relationship of water clarity to various physical and chemical factors has been discussed elsewhere (see Trophic Classification — Factors Affecting Water Clarity). An unusual association is the relationship of water clarity and pH, which were positively correlated in impoundments and negatively correlated in lakes with measured color levels less than 40 units. Color and pH were negatively correlated in impoundments as were color and water clarity, thus pH and water clarity were directly related. This is the opposite of findings reported by Kwiatkowski and Roff (1976) for northern Ontario lakes that had a pH below 6.0 units, where a strong inverse relationship was noted. While this inverse relationship appeared in the low color lakes, the mean pH for this group of lakes was 7.1 units (Table 13), which is not significantly different than lakes with high color levels. The significance of these differences is uncertain. The fact that water clarity was strongly correlated with all other parameters in the low color data subset indicates that color may severely interfere with the interpretation of results among the other data subsets.

Fairly consistent associations of water clarity and turbidity, nitrogen and phosphorus were common to all regions. Water clarity was negatively correlated with calcium in the Southeast and Central regions and with alkalinity in the Southeast Region. This is consistent with other studies which have demonstrated that excess calcium in the form of colloidal particles may affect the penetration of light in the water column (Hutchinson 1975, Kwiatkowski and El-Shaarawi 1977).

CHLOROPHYLL *a*

- − Water clarity
- + Nutrients
  - − Chlorides (impoundments, low alkalinity lakes and low color lakes)
  - − Mean depth (seepage lakes and low alkalinity lakes)
  - − Turbidity (low alkalinity lakes and low chlorophyll *a* lakes)
  - − Maximum depth (low alkalinity lakes and low chlorophyll *a* lakes)

There are several weaknesses inherent with the use and interpretation of chlorophyll *a* data. Chlorophyll *a*:cell volume ratios differ among algal genera and species, and among the same species under differing environmental conditions or at different times of the year. Despite these problems, chlorophyll *a* gives a general indication of the amount of algal biomass present in the lake water at the time of sampling. This value may or may not be representative of a lake's average summer algal concentration due to fluctuations common in phytoplankton biomass. Nevertheless, certain relationships with other parameters (representing existing conditions coincidental to collection of the chlorophyll *a* sample) are evident. In addition to the Secchi-chlorophyll *a* and chlorophyll *a*-total phosphorus relationships which have been previously discussed (Trophic Classification — Factors Affecting Lake Trophic Status), there are a number of other relationships. Among these are the direct correlation of chlorophyll *a* and chlorides in low alkalinity lakes, low color lakes and impoundments. Nitrogen and turbidity appeared to be better correlated with chlorophyll *a* than with total phosphorus in low chlorophyll *a* lakes. A weak association was noted between chlorophyll *a* and alkalinity in low alkalinity lakes.

Regional comparison of chlorophyll *a* with other characteristics showed similar relationships, except for the negative correlation of chlorophyll *a* with magnesium, pH and alkalinity in the Southeast Region. The reason for this difference is unknown but may be related to higher magnesium levels in this region (Fig. 22) or to the impact of the macrophyte communities.

CHLORIDES

- + Calcium
- + Magnesium
- + pH
- + Alkalinity
- + Nutrients
  - + Turbidity (seasonal data, impoundments and low color lakes)
  - − Mean and maximum depth (impoundments)
  - + Color (low alkalinity lakes)

Chlorides were generally associated with nutrients and other ions. The association with color in low alkalinity lakes is unexpected and cannot be explained by us. As previously indicated, chlorides were negatively correlated with depth in impoundments.

CALCIUM

- + Chlorides
- + Magnesium
- + pH
- + Alkalinity
- + Nutrients
  - + Turbidity (seasonal data)
  - − Mean depth (impoundments)
  - + Color (low alkalinity lakes)

Calcium shows a strong positive correlation with almost all other water quality parameters; a notable exception is in low alkalinity lakes where the relationship of calcium with organic and total nitrogen falls off. However, increasing calcium levels generally accompany increasing levels of inorganic and total nitrogen (Append. A).

The regional comparisons again show the Southeast Region to be slightly different than the others in that calcium and pH were negatively correlated.

MAGNESIUM

+ Calcium
+ pH
+ Alkalinity
+ Nutrients
  - – Shoreline development factor (seasonal data and impoundments)
  - + Turbidity (seasonal data and impoundments)
  - – Mean depth (impoundments)
  - – Water clarity (low color lakes)

As would be expected, correlations between magnesium and other parameters were similar to those for calcium. Most associations were significantly positive, except where negative correlations with organic and total nitrogen were found in the Southeast Region.

Considerable overlap in mean magnesium content exists between lakes with low and high levels of phosphorus, but lakes with medium phosphorus content have significantly lower mean concentrations of magnesium than high phosphorus lakes (Append. A). This apparent "dip", which was also noted for calcium, was repeated in the magnesium-inorganic nitrogen relationship. Highest magnesium levels (19 mg/l) were found in lakes with high inorganic nitrogen. Lakes with total nitrogen greater than 1 mg/l had higher magnesium levels than lakes with low total nitrogen.

pH

+ Chlorides
+ Calcium
+ Magnesium
+ Alkalinity
+ Turbidity
  - + Phosphorus (random lakes)
  - – Color (impoundments and drainage lakes)
  - + Water clarity (impoundments and low color lakes)

Generally, pH correlations were highly related to alkalinity conditions. A significant difference was found in the pH of lakes which had low levels of total phosphorus and inorganic and to-

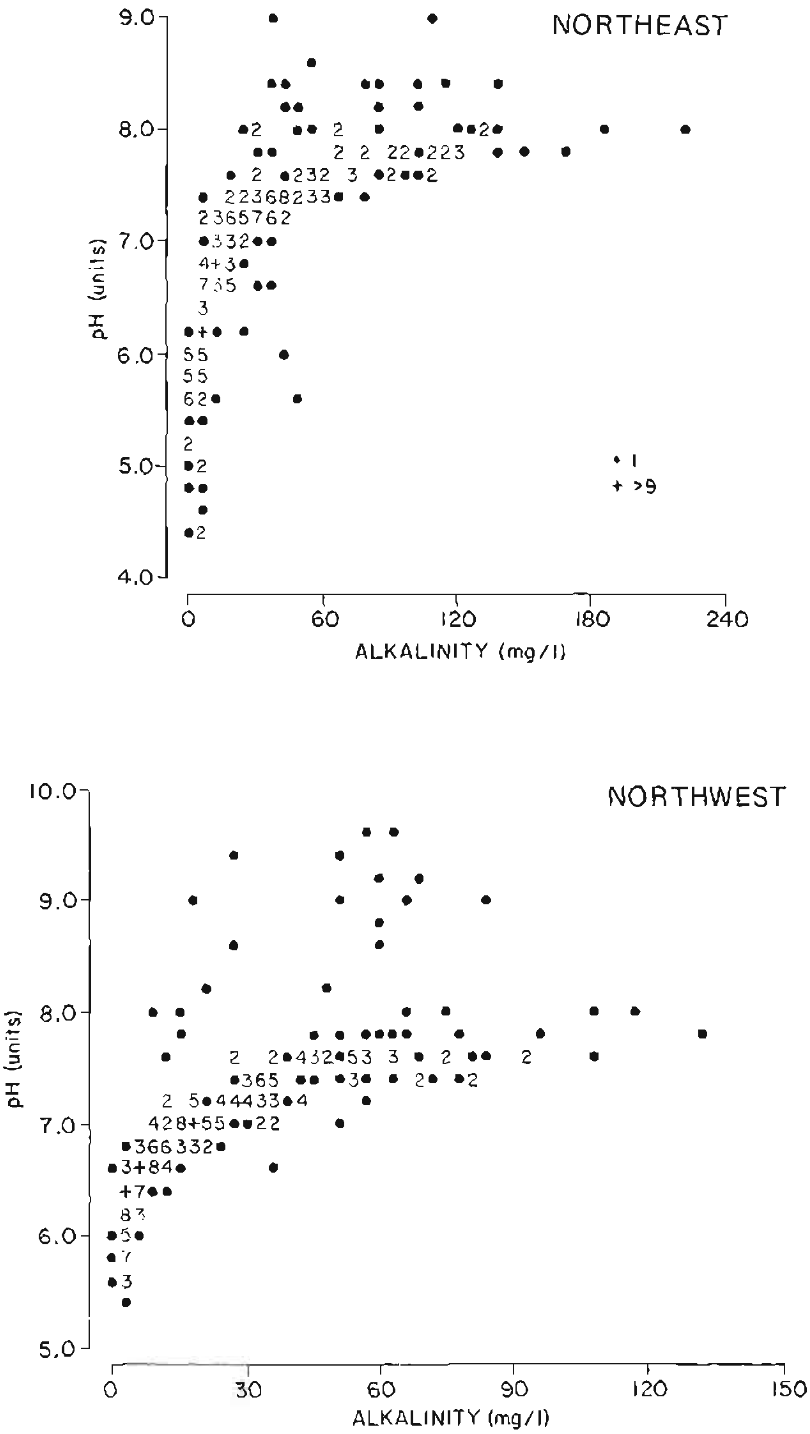
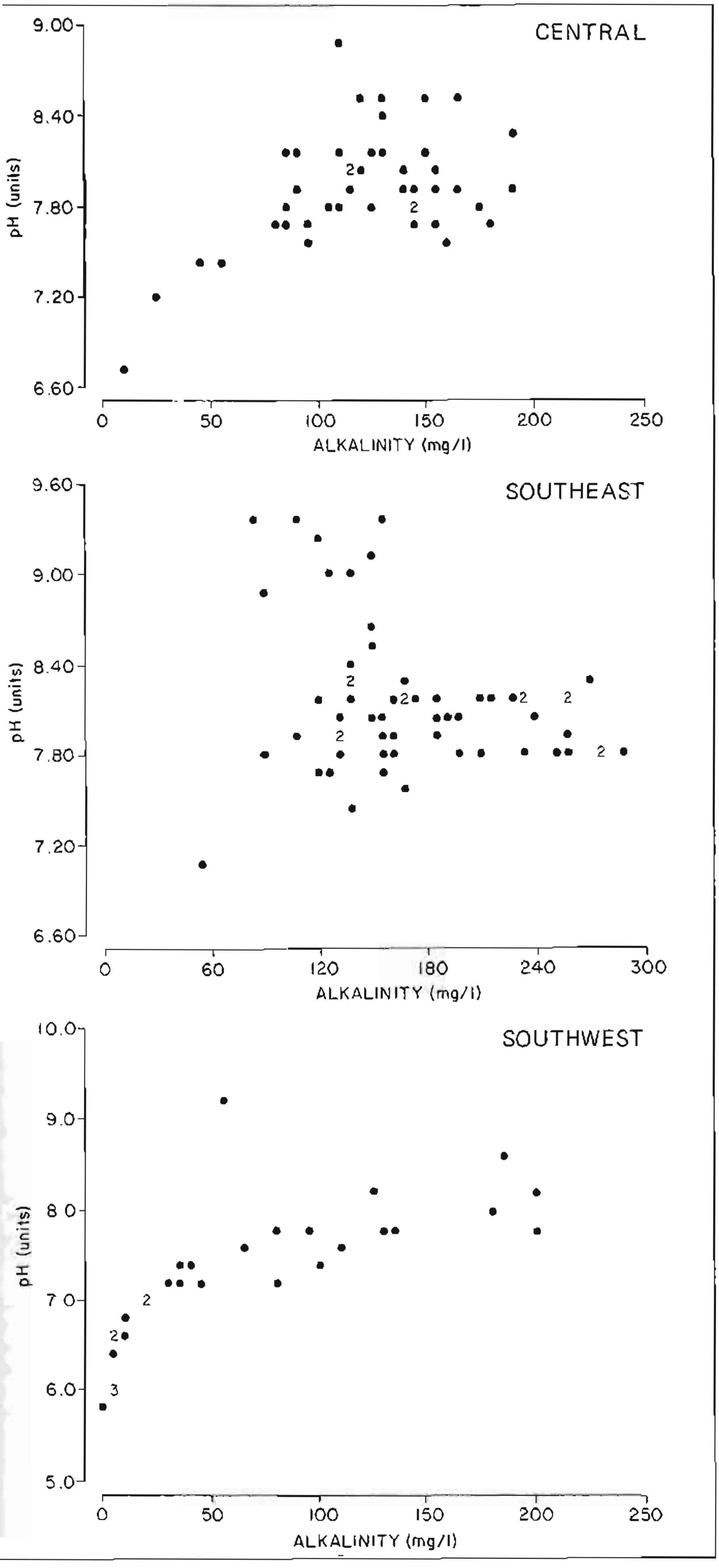


FIGURE 71. *Relationship of pH and alkalinity in five Wisconsin regions (random data set).*



tal nitrogen vs those lakes with high levels (Append. A). The correlation of pH with nitrogen and phosphorus was poorer in impoundments and drainage lakes than in seepage lakes. Whether this was influenced by the generally shorter retention time of impoundments and drainage lakes or the differences in other characteristics in these types of lakes is uncertain. The negative correlation of pH and color was previously discussed.

Of regional significance was the relatively poor correlation of pH with nitrogen and phosphorus in the Northeast, Central and Southwest regions, as opposed to the strong correlations evident in the Northwest Region and the slightly negative correlation with total phosphorus in the Southeast Region. Summaries of the regional data (Table 39) and plots of pH vs alkalinity for each of the regions (Fig. 71) provide a clearer picture of the interrelationship of these two parameters. The Southeast Region was different than the others with an apparent negative relationship of pH to alkalinity. The low correlation (negative) between pH and alkalinity in the Southeast Region appears to be a statistical anomaly created by the 8-10 lakes with pH values above 8.5 and the lack of low alkalinity-low pH lakes in the region. The Northwest Region had a similar number of high pH lakes (although at lower alkalinities) but also had enough low pH-low alkalinity lakes to provide a stronger positive correlation. Most of the high pH lakes in the Northwest Region were located in Polk County where sampling followed a period of warm, humid weather. Algal blooms were evident in many of the lakes, but no correlation was found to link high pH with chlorophyll *a* or nutrient concentration.

It is quite obvious that the pH-alkalinity relationship is nonlinear in form (Fig. 71); thus, discussion of the relationship based on correlation coefficients alone is of questionable value. The decrease in pH with alkalinity becomes very steep in the lower alkalinity ranges, but scatter in pH values changes very little. This relationship is similar to the findings of other investigators and is important in showing the natural variability between the two parameters.

ALKALINITY

- \+ Chlorides
- \+ Calcium
- \+ Magnesium
- \+ pH
- \+ Nutrients
  - − Shoreline development factor (seasonal data and impoundments)

TABLE 39. *The relationship between pH and total alkalinity in five Wisconsin regions.*

	pH-Alk	Median		Minimum		Maximum		Mean		SD		N		
Area	r	pH	Alk	pH	Alk	pH	Alk	pH	Alk	pH	Alk	pH	Alk	R^2
Northeast Region	0.679	7.1	22	4.3	1	8.9	224	6.9	37	0.89	40	243	243	46%
Northwest Region	0.717	7.0	18	5.4	1	9.6	133	7.0	27	0.71	25	282	282	51%
Central Region	0.491	7.9	124	6.7	12	8.9	190	7.9	122	0.38	40	44	44	24%
Southeast Region	-0.261	8.0	160	7.1	51	9.4	290	8.1	173	0.48	55	61	61	7%
Southwest Region	0.738	7.2	42	5.7	2	9.2	202	7.2	67	0.81	65	30	30	54%

- \+ Turbidity (seasonal data, impoundments and low alkalinity lakes)
- − Mean depth (impoundments)
- \+ Area (low alkalinity lakes)
- − Water clarity (low color lakes)

While alkalinity was directly related to nutrients and other ions in Wisconsin lake waters, its relationship with chlorophyll *a* was weak (best in impoundments where a few very high values had great significance, and in low alkalinity lakes). The relationship between alkalinity and water clarity was best in lakes with low color. Alkalinity is sometimes used as a rough indicator of lake trophic status, but it is not a very accurate or reliable one for Wisconsin lakes.

Statewide, the relationships between alkalinity and chlorophyll *a* and total phosphorus were not significant in low chlorophyll *a* lakes. Alkalinity and total nitrogen were significantly related (95% C.I.) in these lakes, and chlorophyll *a* was more highly related to total nitrogen than to total phosphorus. Extensive evaluation of chlorophyll *a* and alkalinity relationships (plots) gave no indication as to what factors caused the poor relationships between the two parameters.

Regional distinctions included a strong positive correlation between alkalinity and chlorophyll *a* in the Northwest and Southeast regions. The reason why these regions are different is not known, but it could be related to higher iron levels in the northern lakes or precipitation of phosphorus with calcium carbonate in the southern lakes (see Gessner 1939).

TURBIDITY

- − Mean depth
- − Water clarity
- \+ pH
- \+ Nutrients
  - \+ Calcium (random data, seasonal data, impoundments and low color lakes)
  - − Maximum depth (seasonal data, seepage lakes, drainage lakes and low alkalinity lakes)
  - \+ Chlorides (seasonal data, impoundments and low color lakes)
  - \+ Magnesium (seasonal data, impoundments and low color lakes)
  - \+ Alkalinity (seasonal data, impoundments, low alkalinity lakes and low color lakes)
  - \+ Chlorophyll *a* (impoundments, low alkalinity lakes and low chlorophyll *a* lakes)

Turbidity was generally positively related to all other characteristics, except for the negative relationships with water clarity and depth.

Mean turbidity levels increased with increases in total phosphorus and inorganic and total nitrogen (Append. A). Relationships between turbidity and nitrogen and phosphorus are undoubtedly due to a number of complicated and interrelated factors, including land surface runoff to lakes and the resuspension of lake bottom material.

Regional comparisons revealed little in the way of significant differences.

ORGANIC NITROGEN

- − Mean depth
- − Maximum depth
- − Water clarity
- \+ Chlorophyll *a*
- \+ Chlorides
- \+ Calcium
- \+ Magnesium
- \+ Alkalinity
- \+ Turbidity
- \+ Other nutrients
  - \+ pH (natural lakes and low color lakes)

Organic nitrogen was highly related to most other lake characteristics. Organic nitrogen levels were considerably higher in lakes with total phosphorus greater than 0.03 mg/l (Append. A). Low organic nitrogen levels were associated with low inorganic nitrogen and low total nitrogen levels (Append. A). In low alkalinity lakes, the relationships between organic nitrogen and chlorides, calcium, magnesium and pH were weaker, which may be an artifact created by lower variations in these parameters in low alkalinity lakes.

Regional relationships of organic nitrogen showed strong correlations with chlorophyll *a* in the Northwest, Northeast and Southeast regions, but not in the Central and Southwest regions. The apparent poorer correlations in the latter two regions may not be real and could be caused by the low range of values in the Central Region and high flushing rates and light limitation in the impoundments of the Southwest Region.

TOTAL NITROGEN

- − Mean depth
- − Maximum depth
- − Water clarity
- \+ Chlorophyll *a*
- \+ Chlorides
- \+ Calcium
- \+ Magnesium
- \+ Alkalinity
- \+ Turbidity
- \+ Other nutrients
  - \+ pH (low color lakes)

Total nitrogen concentrations in Wisconsin lake waters correspond quite well to other lake characteristics. Some important total nitrogen associations are shown in the comparisons of lakes with differing levels of total phosphorus and organic nitrogen (Append. A). Total nitrogen appears to be a fairly good indicator of overall lake water quality, but variability makes delineations of trophic index levels difficult (Fig. 72).

Total nitrogen was not well related to chlorophyll *a* levels in either the Central or Southwest Region, but it was related to chloride levels. As would be expected, these relationships are similar to those noted for organic nitrogen.

INORGANIC PHOSPHORUS

- − Water clarity
- \+ Chlorophyll *a*
- \+ Chlorides

+ Calcium
+ Magnesium
+ Alkalinity
+ Turbidity
+ Other nutrients
    + Area (random data)
    − Mean depth (seasonal data, drainage lakes and low color lakes)
    − Maximum depth (seasonal data and impoundments)

Inorganic phosphorus was positively correlated with other lake characteristics. The weakest correlation was with pH, particularly in impoundments and drainage lakes. A clear relationship exists between inorganic phosphorus and total phosphorus, and inorganic nitrogen and total nitrogen (Append. A).

Significant correlations with all parameters were evident in the Northwest Region, while in the other regions the relationships between inorganic phosphorus and chlorides, magnesium, pH, and in some cases alkalinity were not as good.

TOTAL PHOSPHORUS

− Mean depth
− Water clarity
+ Chlorophyll *a*
+ Chlorides
+ Calcium
+ Magnesium
+ Alkalinity
+ Turbidity
+ Other nutrients
    + Area (random data)
    − Maximum depth (seasonal data and drainage lakes)

Generally, total phosphorus was directly correlated with other water quality determinants; most of these have been discussed previously under other headings and in the section of the report discussing factors affecting nutrient concentrations (Trophic Classification — Factors Affecting Nutrient Concentrations). Phosphorus and nitrogen appear to be highly co-associated (Append. A). The fact that the weakest total phosphorus correlation appears to be with pH, especially in impoundments and drainage lakes, may support other evidence that a lake's watershed has a significant impact upon lake pH.

Strong positive correlations between total phosphorus and all other characteristics were found in the Northwest Region, while correlations were poorer with magnesium, pH and alkalinity in other regions. A negative correlation between total phosphorus and pH was found in the Southeast Region, indicating possible sedimentation or precipitation of phosphorus with rising levels of alkalinity (primarily due to high magnesium levels). Many of

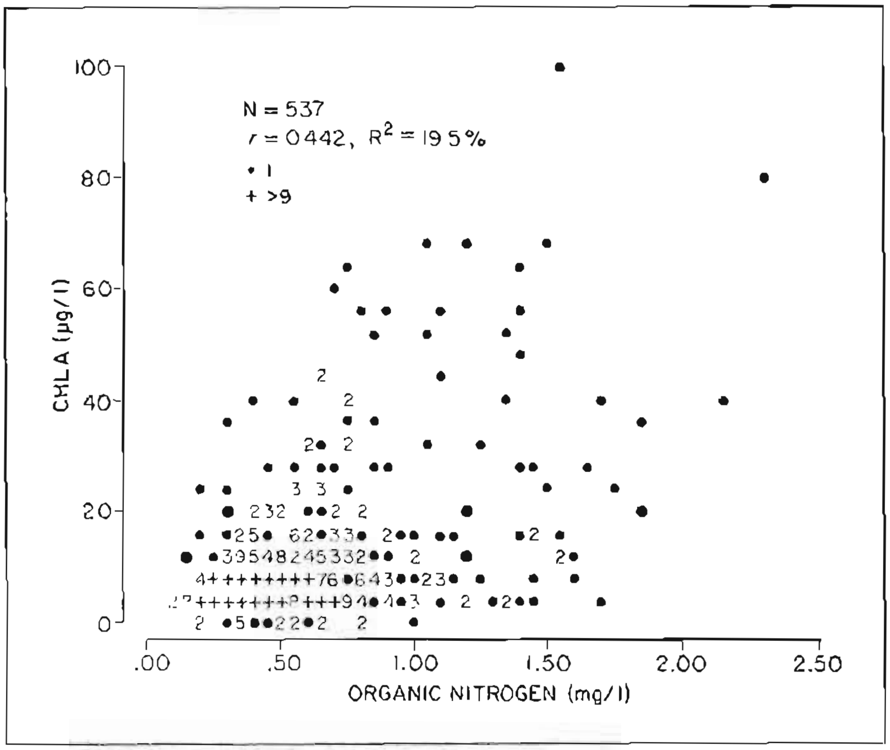


FIGURE 72. *Relationship of chlorophyll a to summer organic nitrogen levels (natural lakes, random data set).*

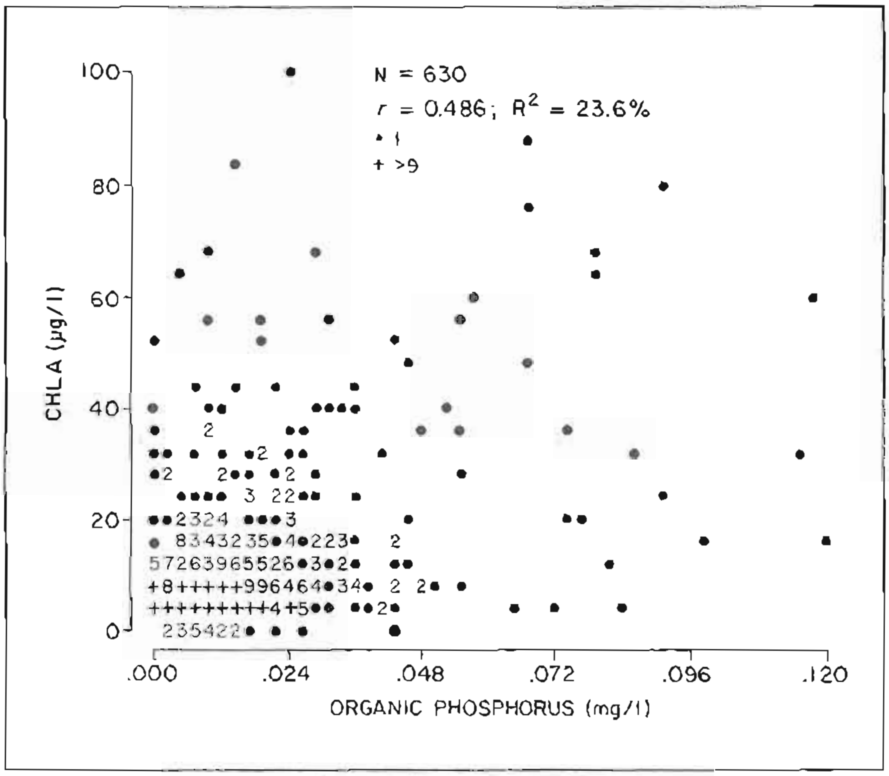


FIGURE 73. *Relationship of chlorophyll a to organic phosphorus (lakes and impoundments, random data set).*

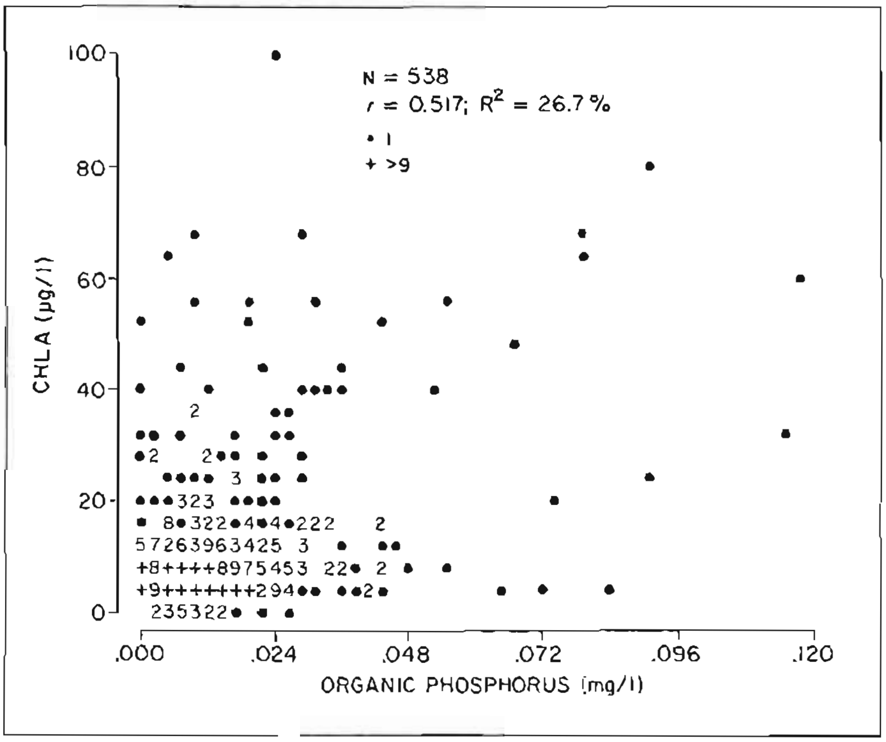


FIGURE 74. *Relationship of chlorophyll a to organic phosphorus (natural lakes, random data set).*

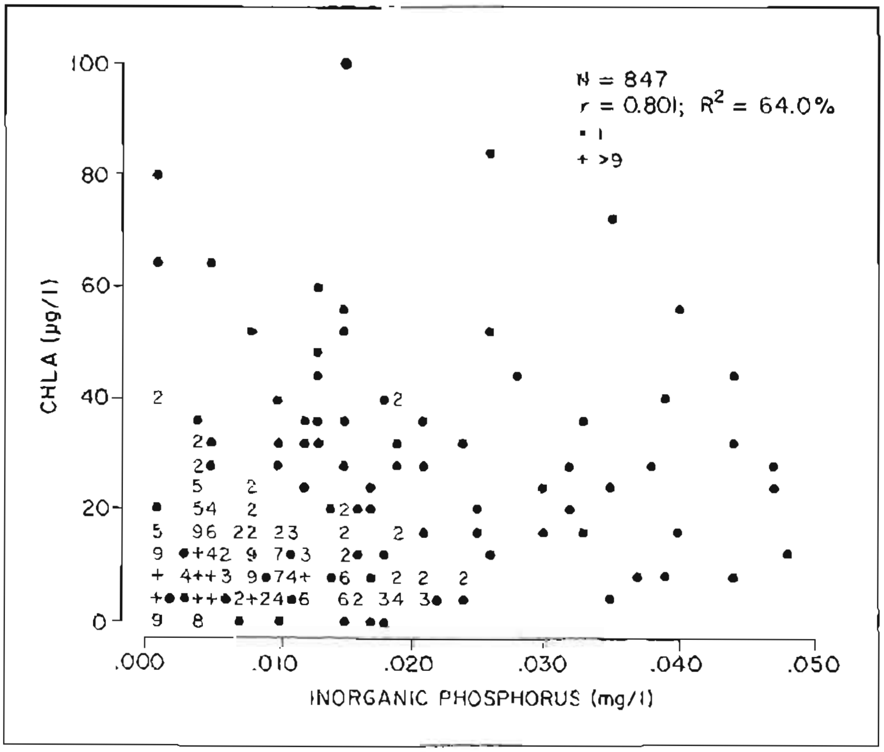


FIGURE 75. *Relationship of chlorophyll a to inorganic phosphorus (natural lakes, total data set). Note: plot is restricted to chlorophyll a values <100µg/l and Inorganic P values < 0.05 mg/l.*

the lakes in this region are large and deep with high hydraulic loadings. The overall trophic condition or water quality of southeast Wisconsin lakes, in view of the high phosphorus loading from their watersheds, is generally better than might be expected. The reasons for this apparent anomaly are unclear, but may be attributed to a combination of factors including the physical morphometry and higher magnesium levels in southeast Wisconsin lakes.

DISCUSSION AND SUMMARY

The interrelationships expressed in Appendix B are undoubtedly masked in many instances by the composition of the lakes in the particular subset (and their associated characteristics). Thus, many relationships may be "hidden" by other overriding factors. Likewise, some of the relationships expressed as significant in Appendix B may purely be the result of their nonrandom distribution (speaking in terms of parameter values not geographic distribution). The relationships and correlations are based on the assumptions that the data are normally distributed and indeed linear. Figure 72, a plot of chlorophyll *a* vs organic nitrogen for the random survey data, shows that while the correlation coefficient ($r = 0.442$; $R^2 = 19.5\%$) is significant ($P > 0.001$), the scatter and skewedness in the relationship are great. A plot of chlorophyll *a* vs organic phosphorus for the random data set illustrates a similar problem (Fig. 73). Reducing the data base to natural lakes only (Fig. 74) improves the correlation, but does little to improve the value of the relationship in terms of its predictive capabilities. Even associations with higher coefficients of determination (R^2) display a great deal of scatter. Figures 75 and 45 demonstrate the weaknesses of correlation coefficients for purposes other than gross generalizations concerning the relationships between characteristics. While R^2 values were very high, chlorophyll *a* ranged from very low to high within different ranges of both inorganic and total phosphorus values. Incorporation of a third factor in the analysis might help explain some of the scatter in many of the relationships. For example, some evidence supporting nitrogen limitation in natural lakes is given in Figure 45. Elimination of low nitrogen:phosphorus ratio lakes would undoubtedly improve this relationship.

A number of the relationships are definitely nonlinear, in which case transformations of the data would probably result in improved correlations (examples are Figs. 76, 77 and 37). Restrictions on data sets also present problems. Particularly noteworthy is the bias interjected by restricting the data set to low chlorophyll *a* lakes and the apparent improvement in R^2 values (Fig. 77). The R^2 for the same relationship for all data was only 8.7%.

Determining which transformation to use to obtain the highest correlations can be a time-consuming task and quite often is academic, since a great deal of scatter remains regardless of the transformation made and transformations may do little to improve the predictive ability of the resulting linear regression analysis.

Log-log transformations, often used in this effort, reduce the importance of large values and quite often succeed in "normalizing" the data. This method often failed to improve the predictive value of the resulting equations because the basic relationship between the parameters is not affected. Log-log transformations of selected relationships vary in their impact on the correlation coefficients (Table 40). The log transformation had little effect on the total phosphorus-chlorophyll *a* and water clarity-color relationships, while R^2s increased in other cases. For our data set, the enormous number of possible transformations of the data, combined with the large number of data parameters, prohibits detailed discussion of many of the interrelationships.

TABLE 40. *Correlation coefficients (r) for selected parameters, Log_{10} transformed (above) and not transformed (below)(random data set, summer data).*

	Log_{10} Chlorophyll *a*	Log_{10} Total P	Log_{10} Secchi Disc Depth
Log_{10} Chlorophyll *a* (µg/l)	—		
Log_{10} Total P (mg/l)	0.568	—	
Log_{10} Secchi disc depth (m)	-0.731	-0.570	—
Log_{10} Color	0.347	0.192	-0.540
	Chlorophyll *a*	**Total P**	**Secchi Disc Depth**
Chlorophyll *a* (µg/l)	—		
Total P (mg/l)	0.582	—	
Secchi disc depth (m)	-0.271	-0.287	—
Color	0.054	0.047	-0.445

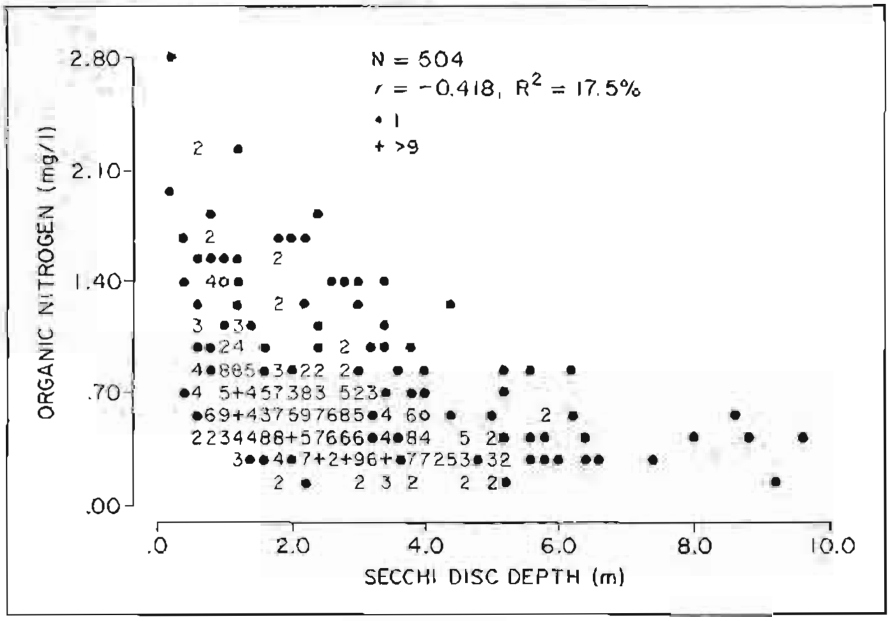


FIGURE 76. *Relationship of organic nitrogen to Secchi disc (natural lakes, random data set). (middle)*

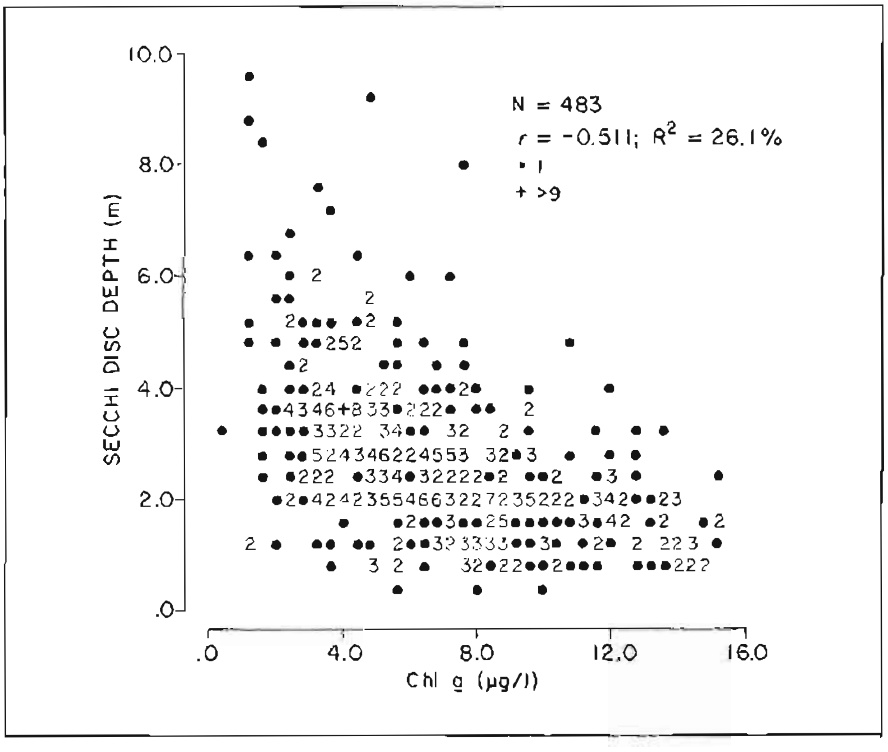


FIGURE 77. *Relationship of chlorophyll a to Secchi disc (natural lakes, low chlorophyll a levels, random data set). (bottom)*

State Historical Society of Wisconsin

The old and the new in sampling of Wisconsin lakes: Water quality information on some lakes spans several decades, but in general there is not enough data to determine trends (above, E. A. Birge at Trout Lake in the 1920's and below, authors collecting samples in the late 1970's).

HISTORICAL TRENDS IN LAKE WATER QUALITY

We assembled and examined historical data on Wisconsin lakes in conjunction with our data in order to provide a reference source for future investigations and to note whatever trends appeared to be developing. There are several recognized problems involved in positively identifying water quality trends in lakes, even though there are probably more historical water quality data on Wisconsin lakes than exist in any other state. Despite this data base, long-term chemical and biological changes in water quality are very difficult and, in many instances, impossible to ascertain (Stewart 1976, Lehner et al. 1980, Lillie and Mason 1980). The reason for this in many cases is that data have not been collected regularly enough to allow for differentiation between long-term trends and normal annual, seasonal, and diurnal variability. Furthermore, differences in sample collection and/or analysis often make comparisons of data inappropriate. In addition, the imprecision of some laboratory procedures results in wide ranges of values, thereby precluding detection of subtle water quality changes.

Notable exceptions to these general observations regarding historical trends are found in those few lakes in the state that have been directly impacted by sewage discharges. Examples of these are Snake Lake, Vilas County, and the Yahara chain of lakes, Dane County. After a treatment plant for the Village of Woodruff began discharging to Snake Lake in 1942, significant increases in nitrogen, phosphorus and chloride were found (Mackenthun 1952). The impact of sewage effluent on the Yahara lakes and the changes that occurred when discharges were eliminated have been documented by Lathrop and Johnson (1979).

We concluded that further detailed statistical comparisons of current and historical water quality data beyond those previously made by other investigators generally were not warranted. We are presenting a synopsis of our historical data assessment and discussing only those water quality indicators on which the most historical data exists. Lakes where the long-term data base is the greatest are examined as best evidence for determining whether or not water quality changes are occurring in Wisconsin lakes.

pH - ALKALINITY

Because of the current interest in the possible impact of acid deposition on the state's water resources, analysis of available pH and alkalinity data on Wisconsin lakes has been made in recent years to determine if changes have taken place (Lehner et al. 1980, Lillie and Mason 1980, Magnuson et al. 1981). Efforts centered on northern soft water lakes, since these are considered to be most susceptible to acid deposition. Magnuson et al. (1981) compared 1979-80 pH and alkalinity measurements of 61 northeastern Wisconsin lakes with 1925-41 values for the same lakes that were taken from the archives at the University of Wisconsin-Madison. Alkalinities in the sampled lakes were in many cases slightly higher in 1979-80 than in 1925-41; pH appeared to be unchanged in lakes where values were between 5.7 and 6.9 during the earlier period, but increased in lakes with early pH's between 7.0 and 8.5. Baker and Magnuson (1976) reported a slight increase in pH and alkalinity of Crystal Lake in northeastern Wisconsin from 1924 to 1973, which was attributed to eutrophication processes.

Even though the pH and alkalinity data base on Wisconsin lakes is relatively large, the data are of limited value due to gaps over time, natural fluctuations, and unanswered questions concerning data comparability. Determinations of pH have been reported to vary considerably depending on method used (colorimetric indicators vs electrometric), especially in waters with low buffering capacity, and alkalinity values can vary because of differences in titration procedures (Galloway et al. 1979, Haines 1980).

The variants of statistical change in lake pH are best demonstrated by the data from four extensively studied Wisconsin lakes. Early published data (Juday, Fred, and Wilson 1924, Birge and Juday 1927, Juday, Birge, and Meloche 1935, Allgeier, Hafford, and Juday 1941, and Hile and Juday 1941) on Trout, Crystal and Nebish lakes, Vilas County, and Devil's Lake, Sauk County, were compared with more recent Department of Natural Resources (DNR) data (DNR 1972, Kempinger and Christenson 1978, and our data). The pH data comparisons did not show any trends as evidenced by linear regression slopes. An important observation to be made from these comparisons is the wide range of values reported for some of the lakes.

The summer pH profile data from each of the four lakes (which all thermally stratify) were combined (by lake, not as a group) and the range, mean, and confidence intervals are shown in Figure 78. Surface pH values for Trout Lake during 7 separate summer observations (6 different years) varied from 7.6 to 8.0 with an average pH value of 7.7. Of primary significance is the decrease in pH with depth and the narrow range of values at 10 m. Trout Lake has a mean surface water alkalinity of 44 mg/l and is the largest (3,816 acres) and deepest (115 ft) of the four lakes observed. The pH and alkalinity of the lake appear to have a narrow range for all seasons when compared to the other three lakes, probably due at least in part to its oligotrophic state and great volume.

The 14 separate sets (14 years) of summer pH profiles for Devil's Lake in Sauk County from 1920-78 also show a significant difference between means of surface and bottom waters. A sharp decrease in pH occurs between the epilimnion and hypolimnion. The range of surface pH values (6.9-8.0) is greater than that found in Trout Lake, which may be due to such factors as smaller size (351 acres), greater fluctuations in water level, and higher trophic level.

In the Nebish Lake summer pH profile data for 9-12 dates, there is a significant difference between surface and bottom pH means and the typical decrease in pH with depth ($P < 0.05$). The variation in the data is apparent from the range of values (6.2-7.7) with a standard deviation about the mean of $\pm$0.49 pH units. Nebish Lake (91 acres) is smaller than both Trout Lake and Devil's Lake and has a lower alkalinity (avg. = 10 mg/l).

Other seasonal sample means are given for comparison in the pH profile data for Crystal Lake (5-7 dates). Crystal Lake, with a mean alkalinity of only 2 mg/l, has an extremely small watershed and is unproductive. It shows the greatest variation in pH data of any of the four lakes — both at the surface (5.5-7.1) and the bottom (5.3-6.2). The profile representing the sample means is nearly vertical, quite different from the others. Also, the pH sample at the surface in winter is higher than the summer value, which is not typical of other lakes.

The pH profile data presented seem to be ample evidence that natural fluctuations can be important and are much greater in some lakes than others.

Such natural pH fluctuations obscure attempts to determine whether or not lake pH changes have occurred over time. That fact is further shown in Figure 79, which gives recent seasonal pH data for Crystal, Trout, and Devil's lakes. Each circle represents the surface pH value from only one date for each season. The data for Crystal Lake illustrate the danger involved in trying to make evaluations of long-term trends with too little data. The 1972-74 data show what appears to be a dramatic drop in pH values from 6.8 to 5.5, followed by a return to 6.8 the following winter. If sampling had ceased with the 5.5 value, it might have seemed reasonable to conclude that a decrease in pH had occurred. However, a check of earlier data produced a surface pH value of 5.5 in the summer of 1932 (Juday, Birge, and

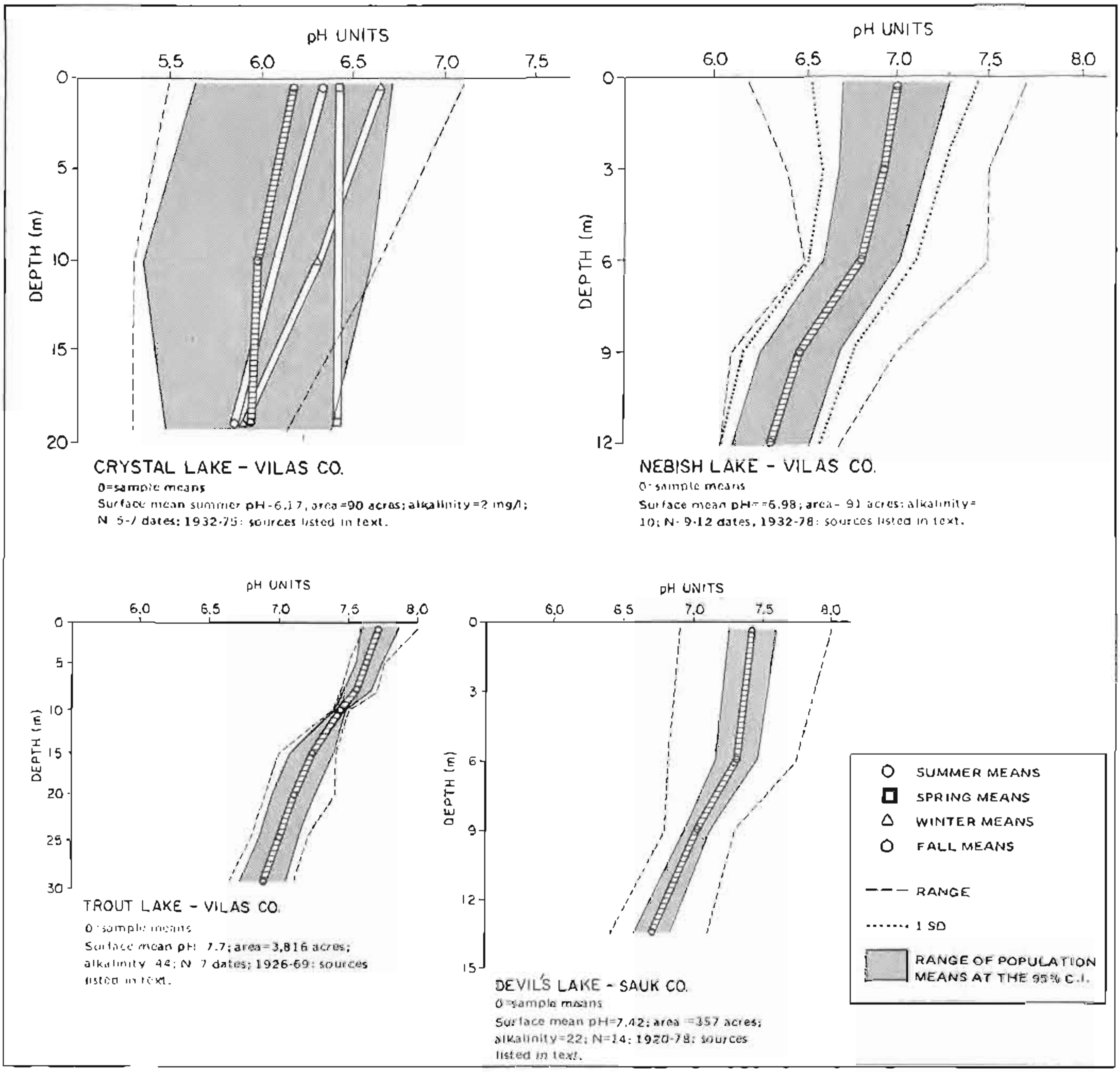


FIGURE 78. *Historical pH profiles for several Wisconsin lakes.*

Meloche 1935) and an extremely low surface value of 4.2 during the winter of 1960 (Poff 1961). Trout Lake pH appears to follow a cycle that repeats itself, while there seems to be no pattern to the pH flux in Devil's Lake, which may be related to weather conditions and water level in the lake. These figures were constructed from discrete data that usually represented only one sampling/lake/season. Sample site locations within lakes can also influence pH measurements. Because of daily and seasonal fluctuations and possibly site location, these data points may not necessarily be representative of the whole lake pH during any given day or season. Daily fluctuations as great as $\pm$ 2.45 pH units have been reported for lake waters (Philip 1927).

Juday, Birge, and Meloche (1935) found annual differences ranging from greater than 0.5 pH unit to greater than 1.0 pH unit in 73.5% and 30%, respectively, of the 245 lakes they studied. The extreme case was a decrease in Adelade Lake (Vilas County) pH from 8.8 to 6.3 between August 1925 and August 1929.

These variations in pH values and the complicated factors bearing on lake pH make it extremely difficult to definitively assess long-term pH trends in Wisconsin's lakes.

Alkalinity was found to have increased in some northeastern Wisconsin lakes (Magnuson et al. 1981). Lehner et al. (1980) also observed that the most significant increases occurred between 1960-62 and 1979. In comparisons of previous alkalinity data on northwestern Wisconsin lakes with our random survey data collected in 1979, this trend was not observed.

Since very little alkalinity data were collected on northwestern Wisconsin lakes by Birge and Juday, the only previous data set available for these lakes (other than miscellaneous file data collected mostly by fish managers) is the Surface Water Resources publications for Northwest District counties (Wis. DNR 1961-78). An important consideration in comparing the Surface Water Resources data and our data sets is that the former were collected at various times of the year during the 1960's and early 1970's, while the random survey data were all collected during the summertime; therefore, the two data sets are not directly comparable. Also, methods used in alkalinity determinations differed somewhat. In the surface water surveys, alkalinities were titrated to either methyl-orange or methyl-purple endpoints around pH

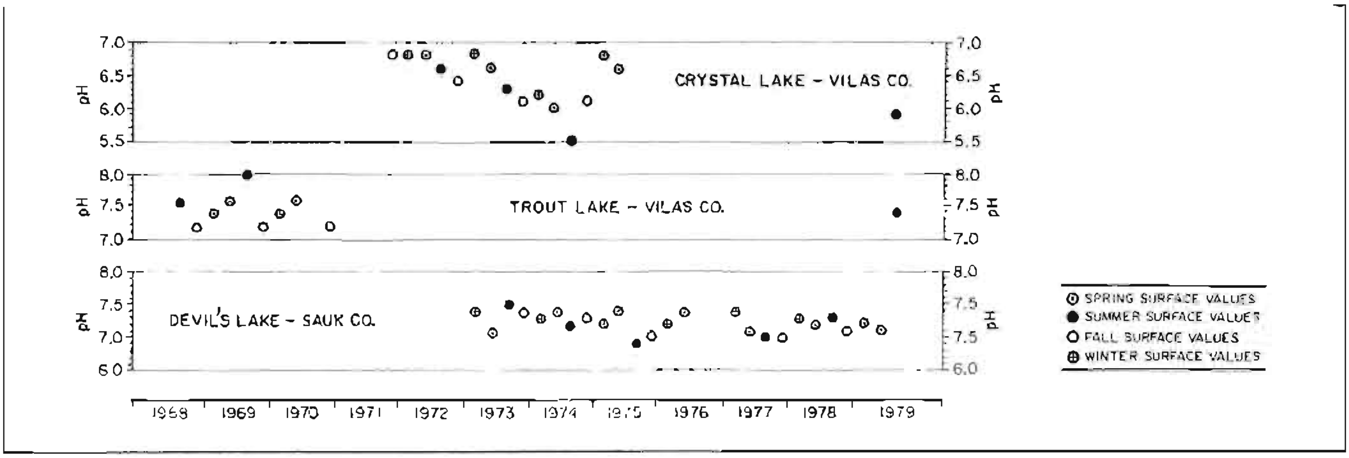


FIGURE 79. *Seasonal and annual variations in pH of three Wisconsin lakes.*

4.7, whereas the random sample alkalinities were titrated potentiometrically to pH 4.5 endpoint.

The distribution, mean, and standard deviation of alkalinities for 256 northwestern Wisconsin lakes sampled in both surveys show no significant differences between data sets (Fig. 80). However, the data comparison did show more lakes where alkalinity appeared to decrease (55%) rather than increase (13%). On this basis it is possible that alkalinity of some northwestern Wisconsin lakes has decreased.

CHLORIDE

Beeton (1965) has reported that chloride concentrations in Lake Michigan (and other Great Lakes) increased during the last century due to cultural effects. There is ample evidence that the same is true for inland lakes in southern Wisconsin. Because of the significant number of lakes of different type and with wide geographical distribution where chloride content is shown to have increased, it appears the trend may be general for lakes in the southern part of the state (Fig. 81).

Trend data on the southern Wisconsin lakes where Birge and Juday measured chloride prior to 1910 indicate relatively little increase from the early 1900's to about 1960 (Fig. 81). Since then, chloride levels appear to have increased at a steady rate, as best exemplified by the Lake Mendota (Dane County) graph, which is based upon the largest data set available. Chloride levels in other important southern Wisconsin lakes have also increased sharply since 1960; even in Big Green Lake (Green Lake County), which because of its great volume would be less likely to show change, chloride concentrations are shown to be increasing. Data for the lower three lakes in the Yahara chain (Monona, Waubesa, Kegonsa), which received sewage efflu-

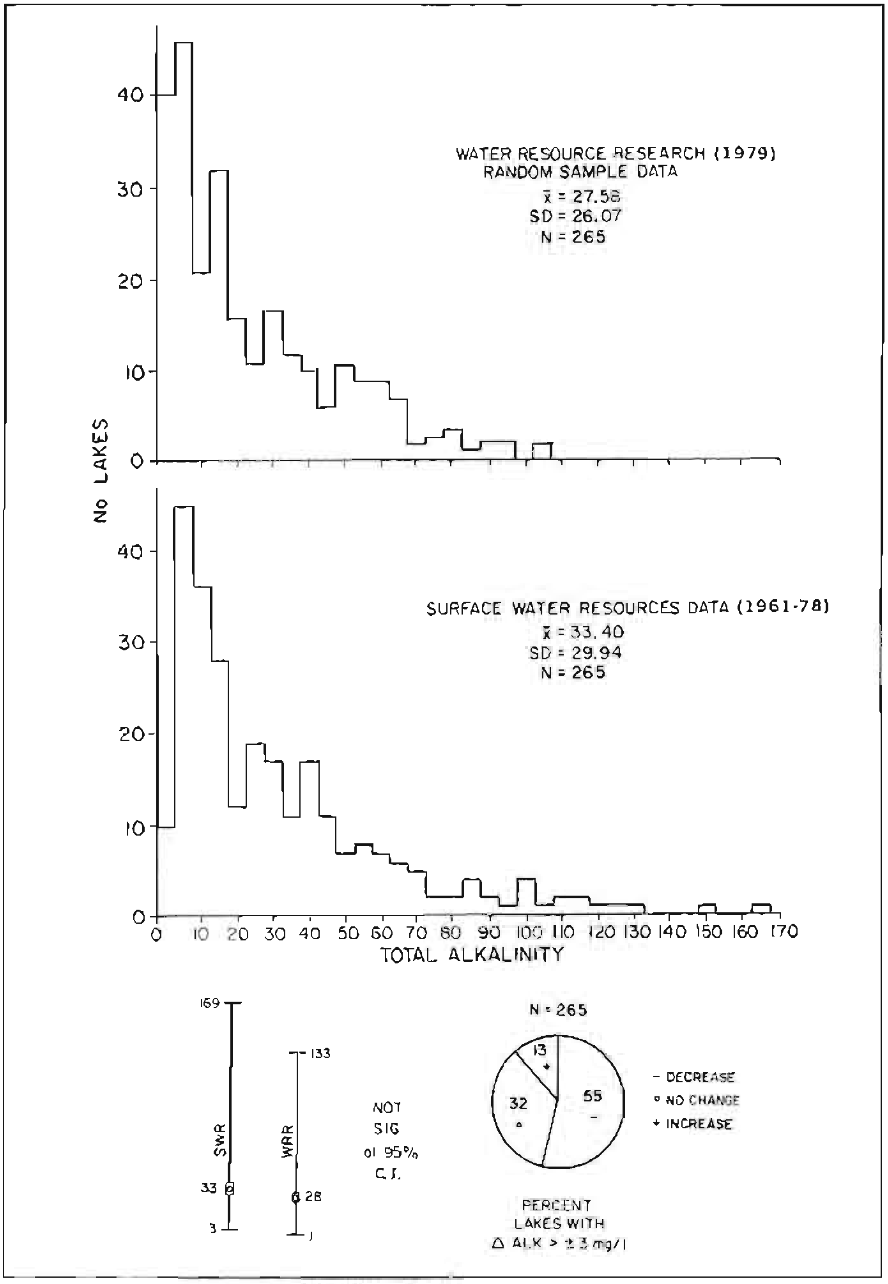


FIGURE 80. *Comparison of alkalinity values in Northwest District lakes.*

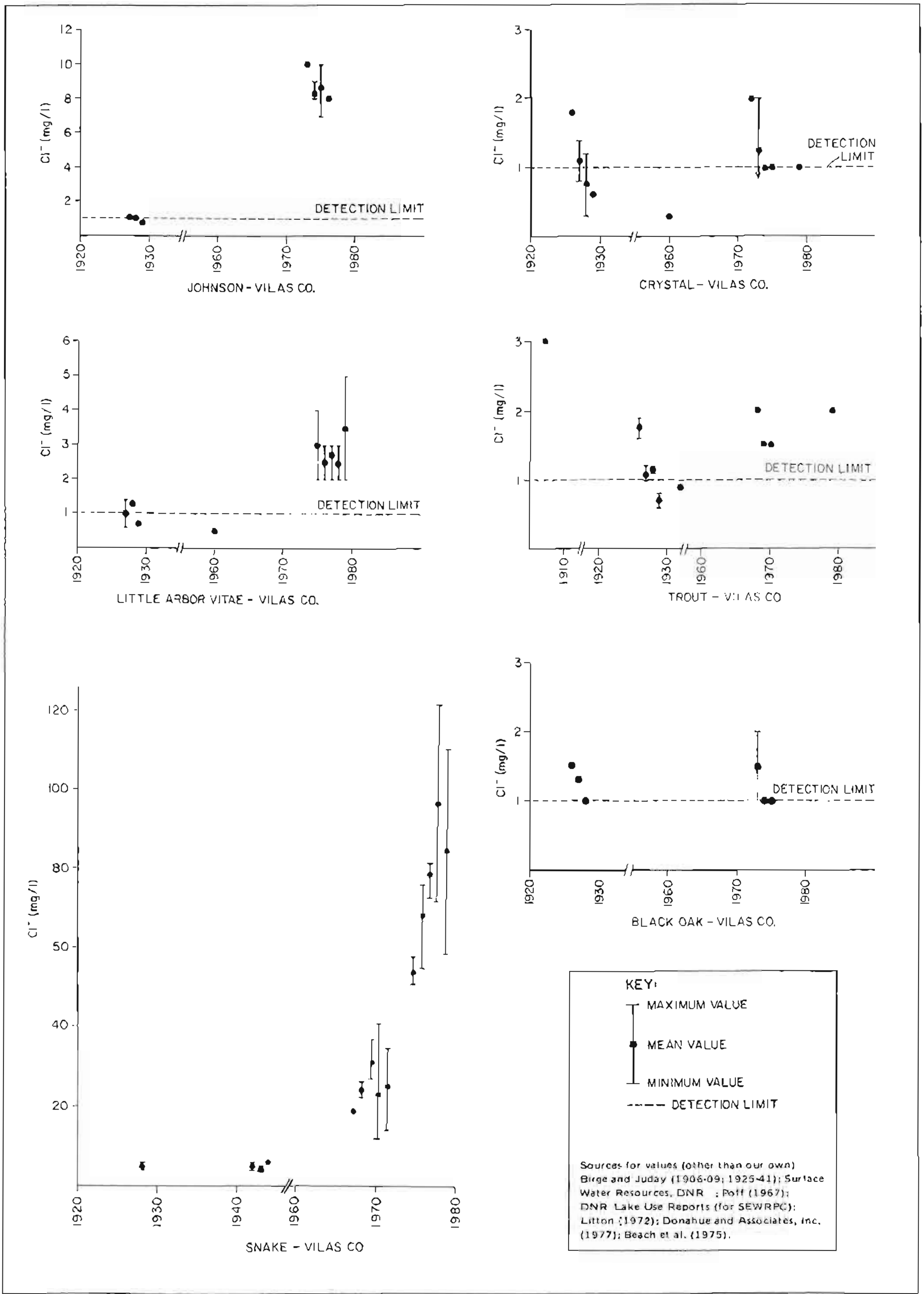


 FIGURE 81. *Historical trends in chloride concentrations in selected Wisconsin lakes.*

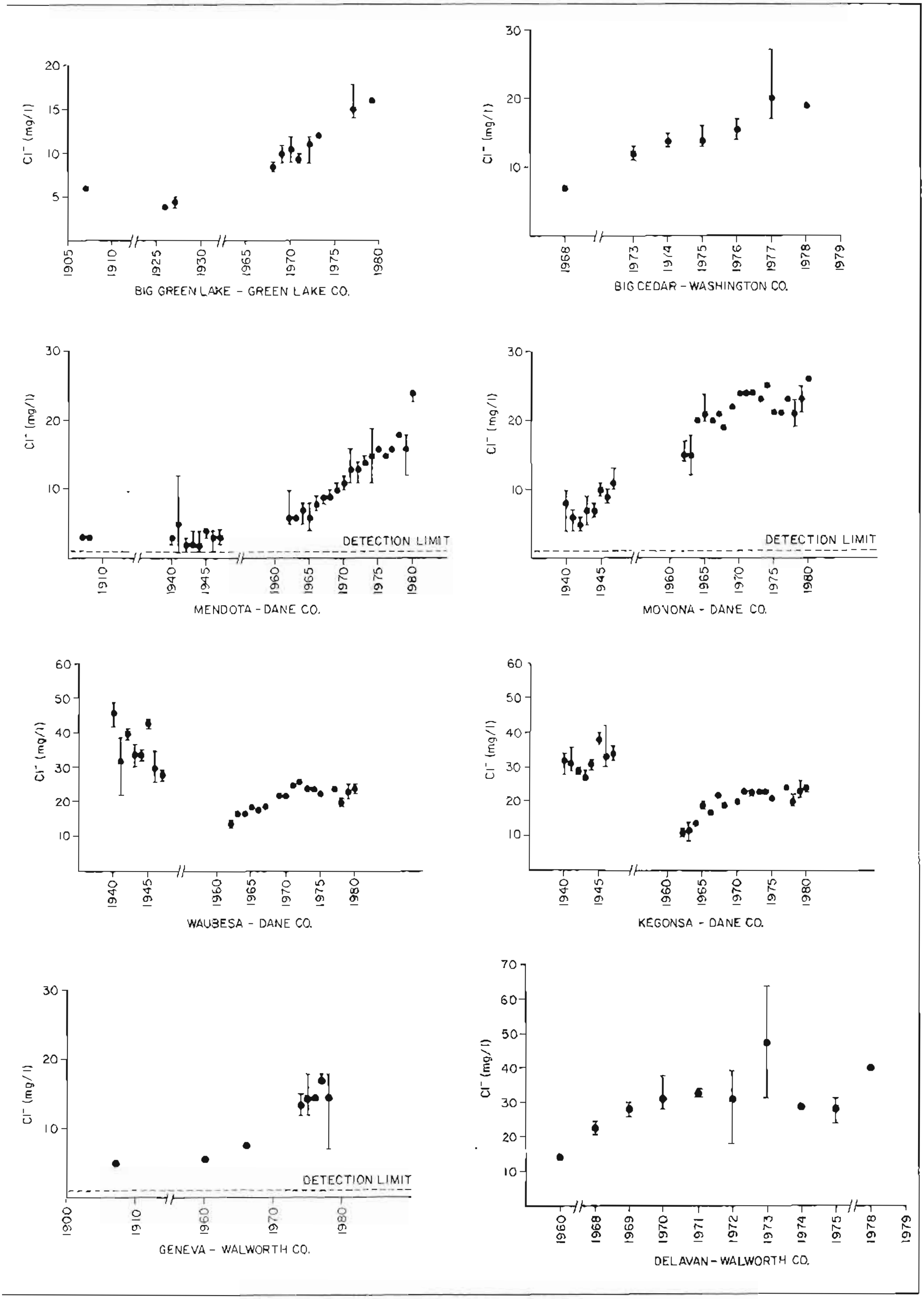


FIGURE 81 (Cont.)

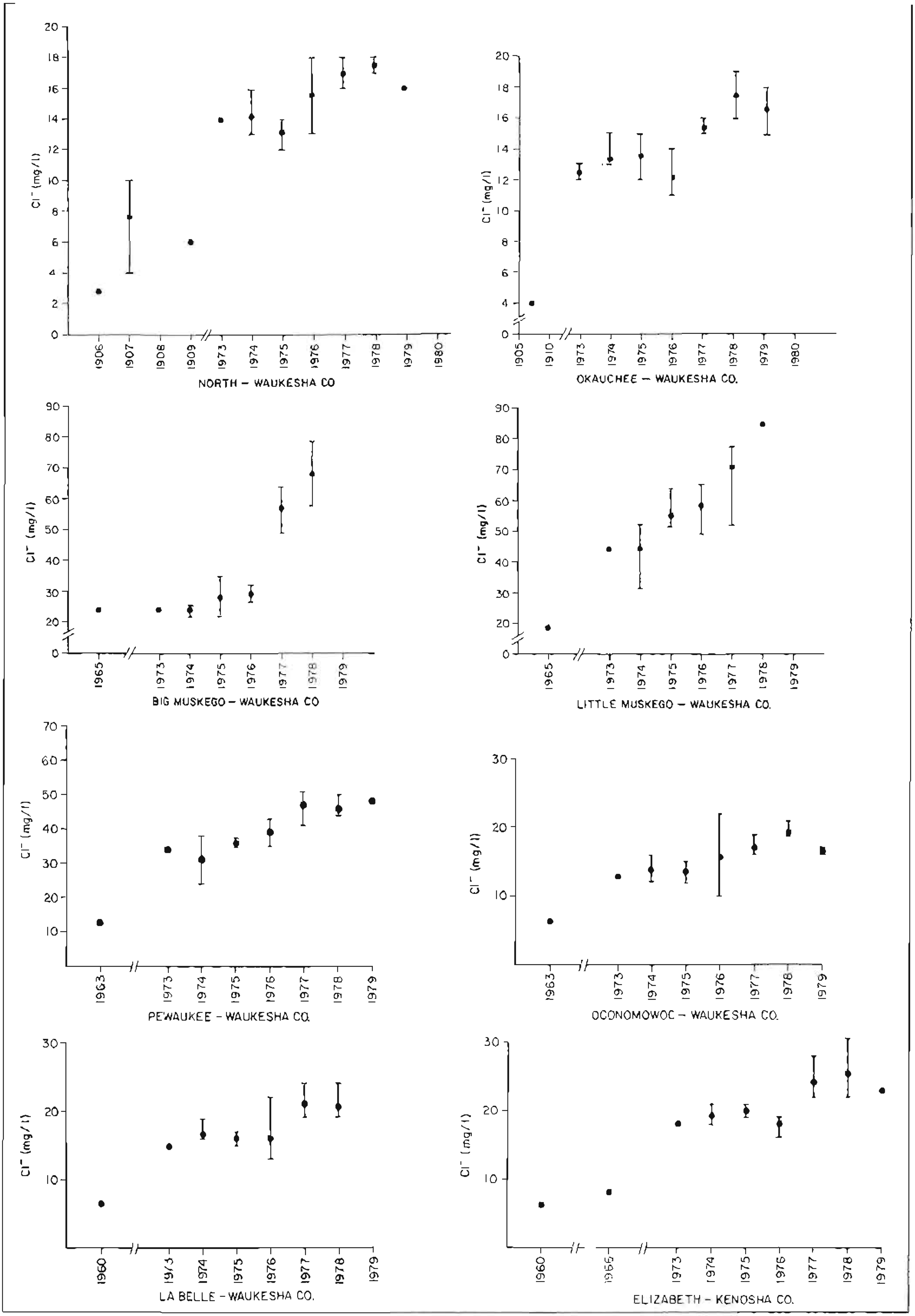


FIGURE 81 (Cont.)

TABLE 41. *Historical sulfate concentrations (summer) in some Wisconsin lakes (mg/l).**

Southern Lakes					Northern Lakes						
Year	Geneva Walworth County	Mendota Dane County	North Waukesha County	Okauchee Waukesha County	Year	Trout Vilas County	Crystal Vilas County	Big Arbor Vitae Vilas County	Black Oak Vilas County	Clear Oneida County	Johnson** Vilas County
1906	13	-	13	-	1907	6	-	-	-	5	-
1907	-	15	14	14	1927	5	2	6	-	3	-
1909	-	-	13	-	1928	3	3	6	5	4	4
1910	-	4	-	-	1969	10	-	-	-	-	-
1948	-	9	-	-	1972	-	2	-	-	-	-
1949	-	9	-	-	1973	-	-	-	2	4	5
1950	-	9	-	-	1974	-	6	-	5	5	3
1966	34	-	-	-	1975	-	-	5	-	-	6
1968	26	-	-	-							
1970	28	-	-	-							
1972	17	-	-	-							
1973	29	-	33	-							
1974	-	-	40	36							
1975	32	18	30	36							
1976	-	21	-	14							
1977	-	25	-	-							
1978	21	14	-	-							

*Data sources: Birge and Juday (1906-10, 1925-41), Lee (1962), present study.
**T40N R6E Sec. 34.

ent from the City of Madison for many years, follow a different pattern. In Lake Monona, the trend toward increase in chloride apparently began in the 1940's, leveled off in the late 1960's and the 1970's, and in the future probably will closely parallel that of Lake Mendota, its primary water source. Chloride concentrations in Lakes Waubesa and Kegonsa dropped significantly between 1947 and 1962, probably due to diversion of sewage from the lakes in 1958 and subsequent flushing by water from the upper lakes with lower chloride content. Since 1962, a trend upward again is evident.

The sources of the chloride accumulating in southern Wisconsin lakes have not been definitely determined, but could include road deicing salt, human and animal wastes, water softeners, and natural deposits or even possibly airborne deposition. Road salt appears to be the most important factor because increases in lake chloride content are closely aligned with rise in use of road salts in recent years. The impact of the increase in chloride on lake ecosystems is unknown, but in itself is probably not significant at present levels. However, chloride has been regarded by some limnologists as a tracer element, and the fact that it is being contributed to lake systems in southern Wisconsin in increasing quantities suggests that contributions of other water quality contaminants, which potentially may be more harmful but cannot be as easily and accurately measured as chloride, may also be increasing.

Chloride content of Vilas County lakes appears to be unchanged (Fig. 81), except for such isolated cases as Snake Lake and Johnson Lake where increases have occurred, probably attributable to road salt and/or domestic wastes. This same situation undoubtedly applies to other areas of northern Wisconsin; chloride content probably has not changed in most lakes but has increased in a few that lie adjacent to major roadways and populated areas.

SULFATE

Trends in sulfate concentrations in Wisconsin lakes are of interest because of the role of sulfate in eutrophication and acidification processes. However, historical sulfate data on Wisconsin lakes is very scanty and inconsistent. Because Beeton (1965) found sulfate levels had increased in all of the Great Lakes except Lake Superior over the past 100 years, it might be expected that sulfate concentrations have also increased in inland lakes, but trends cannot be determined due to the scarcity of data. Table 41 shows summer sulfate levels in three southeastern Wisconsin lakes — Geneva, North and Okauchee — were generally higher between 1966 and 1978 than they were before 1910. However, data are sketchy and great variation in concentrations are evident. Similarly, data for Lake Mendota are inconsistent, although sulfate concentrations are mostly higher in recent years than in 1950 and earlier. The little data available on northern Wisconsin lakes suggest no change in sulfate levels since the early 1900's. An important factor to consider in assessing sulfate data is that until very recently the laboratory test for sulfate was relatively imprecise, which could possibly account for some of the variability apparent in the data in Table 41, and could mask small changes in sulfate at the low levels generally found in northern Wisconsin lakes.

NUTRIENTS

It is a widely held concept that many Wisconsin lakes, particularly in the southern part of the state, probably have become increasingly more eutrophic over the past century due to cultural activities in their watersheds. An increase in levels of plant nutrients, particularly nitrogen and phosphorus, in lake waters is generally believed to be associated with the eutrophication process. While increases in nitrogen and phosphorus may in fact be occurring in some Wisconsin lakes, data are not available showing changes in lake nutrient concentrations, aside from the lakes which have been impacted by point-source sewage discharges.

In the Vilas and Oneida County lakes with the longest historical phosphorus record, great variability but no trends in summer phosphorus levels are apparent (Table 42). The surface (0) values for Trout and Crystal lakes show the wide range in phosphorus concentrations — from 7-42 and 7-48 µg/l, respectively — reported for these two

TABLE 42. *Summer total phosphorus levels (μg/l) in some Vilas and Oneida County lakes.**

Date	Depth (m) 0	5	9	10	15	16	20	22	24	25	30	31	32.5
Trout-Vilas													
25 June 1926	20			20			20				25		
9 July 1926	19												
31 July 1926	15			15			15			18	27		
23 August 1926	22	20		20			20			30		30	
24 June 1927	18												
19 July 1927	17												
20 August 1927	18	20		20	20		20			20		33	
25 July 1928	18												
August 1928	15												
16 July 1929	12												
27 August 1929	10												
28 June 1931	17												
1 July 1931	22												
27 August 1931	13												
22 August 1932	12												
6 June 1966	42											56	
1 July 1966	24											36	
30 August 1966	22											60	
9 July 1968	26		26						13				50
26 August 1969	32				32								32
2 September 1970	20					20							
25 June 1972	7	7						12					
23 August 1972	7	7			9			42					
20 August 1979	19								10				

Date	Depth (m) 0	4	5	10	15	18
Muskellunge-Vilas						
29 June 1926	26					
20 August 1926	20					
2 July 1927	20		20	22	34	34
27 June 1928	17					
19 July 1929	13					
26 August 1931	18					
5 July 1932	22					
25 August 1932	17					
31 July 1940	40					
12 August 1973	30		30			
8 August 1974	30	50				

Date	Depth (m) 0	5	10	12	15	18	21	22	22.5
Tomahawk-Oneida									
29 June 1927	18								
26 July 1927	20	20	22		24	38	40		
8 August 1928	13								
15 August 1973	20				20				20
14 August 1974	20			20				20	
20 August 1979	22								

Date	Depth (m) 0	5	8	10	15	18	20	23	24	25
Black Oak-Vilas										
25 August 1926	19	17		23	22		55	85		
15 August 1927	17		18	20		22			110	
29 August 1928	15									
8 August 1973	30				20					60
9 August 1974	30								30	

Date	Depth (m) 0	5	6	8	9	10	11.5	12	15	18	19.5	21
Crystal-Vilas												
26 June 1926	15	15				15			18	20		
17 August 1926	15					15			15	15		
1 July 1927	14	14				16			18	18		
26 June 1928	12											
21 August 1928	13											
20 August 1929	11											
14 July 1931	12											
6 June 1960	26											
6 June 1966	48									28		
1 July 1966	12									24		
30 August 1966	22									52		
1 June 1972	10		20				20					
25 June 1972	5	5		6								13
1 August 1972	20		20				40					
23 August 1972	7	8		6					8			12
7 August 1974	30							10			10	
22 August 1979	19				10							

*Sources: Birge and Juday (1931), Juday, Birge, Kemmerer and Robinson (1927), Juday and Birge (1925-1941 data unpubl.), Surface Water Resources Series, Poff (1967), Lueschow et al. (1970), EPA (1972), present study.

TABLE 43. *Surface cation concentrations of some southern Wisconsin lakes, early 1900's vs recent times.**

		Sample	Ca		Mg		Na		K	
Lake	County	Dates	Range	Mean	Range	Mean	Range	Mean	Range	Mean
North	Waukesha	1906-09	31-47	37(4)	25-30	28(4)	2.4-3.4	2.9(2)	1.2-1.7	1.5(2)
		1973-79	53-90	63(15)	27-60	39(15)	4-10	6 (18)	0.5-5.7	2.0(18)
Mendota	Dane	1906-10	17-29	24(6)	21-26	23(6)	-	2.7(1)	-	2.6(1)
		1975-79	22-37	30(26)	30-36	32(27)	5-19	8 (24)	2.3-10.8	3.4 (25)
Big Green	Green Lake	1907	-	16(1)	-	26(1)	-	3.0(1)	-	3.1(1)
		1968-72	19-43	32(14)	25-38	36(14)	5.8-14	7.9(16)	2.4-4.8	3.1(16)
Geneva	Walworth	1907	-	18(1)	-	26(1)	-	4.6(1)	-	2.5(1)
		1974-78	26-49	35(33)	26-43	38(33)	4.4-15	8.3(33)	0.9-3.9	2.1(33)
Devil's	Sauk	1908	-	3(1)	-	1(1)	-	-	-	-
		1973-79	3-15	8(21)	3-12	5(21)	-	-	-	-

*Data Sources: 1906-10 - Birge and Juday (unpubl.); 1968-79 - Water Resources Res. Sect. (unpubl.). Number of different sampling dates in parentheses.

lakes. The reason for this variation is not known, but such large and erratic changes in phosphorus concentration in two oligotrophic Vilas County lakes seem unlikely. Instead, differences in the laboratory procedures and results of the various data collectors probably account for most of the variability observed. Similar lack of data and inconsistencies were found in assessing the historical nitrogen data; therefore trends could not be determined.

MAJOR CATIONS

Historical data on major cation concentrations in Wisconsin lakes are very sketchy, but comparison of Birge and Juday data with ours on some important southern Wisconsin lakes strongly suggests that an increase in calcium, magnesium, and sodium has occurred (Table 43). Also, a plot of surface sodium concentrations in Lake Geneva (Fig. 82), although exhibiting a great deal of scatter in the data points, reveals an apparent upward trend since 1966, which seems logical in view of the increase in chloride content of the lake over the past 20 years. Beeton (1965) reported an increase in calcium, sodium, and potassium levels in some of the Great Lakes, further support for the evidence that cations probably are increasing in some southern Wisconsin lakes.

Although Magnuson et al. (1981) found an increase in alkalinity of some northern Wisconsin lakes, which would be expected to indicate higher cation content, the data available do not demonstrate a change. The long-term calcium and magnesium data available on Trout, Crystal and Muskellunge lakes, Vilas County, indicate no significant change in concentration between the Birge and Juday period and 1960-79 (Fig. 83).

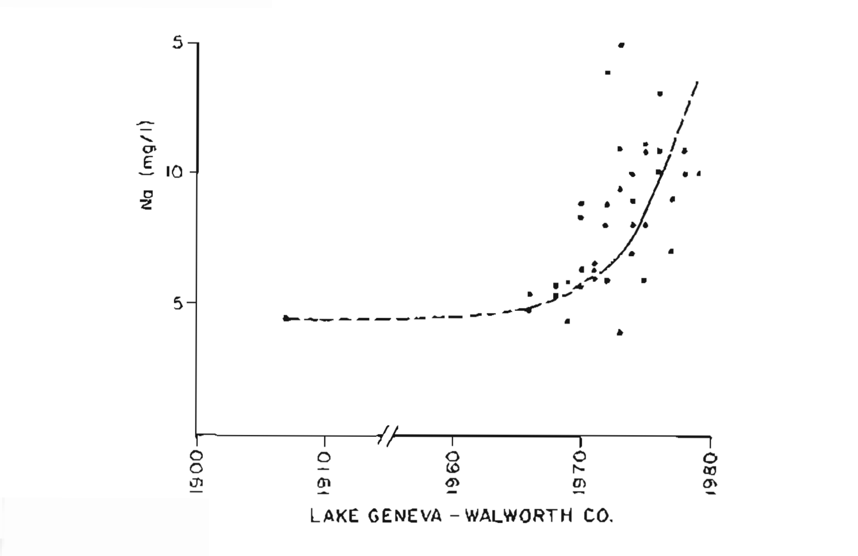


FIGURE 82. *Surface sodium levels in Lake Geneva, Walworth County.*

WATER CLARITY

Water clarity is one of the most important and easily measured water quality indicators, yet it is also one which generally shows great daily, seasonal, and yearly variability, especially in biologically productive lakes. For this reason, long-term changes in water clarity are very difficult to pinpoint. For example, Stewart (1976) analyzed the substantial data set available on Lake Mendota (Dane County), and while noting a possible trend toward reduced water clarity in summer and better clarity in winter, he concluded that the data generally provided little evidence of increasing eutrophication of the lake. In comparing historical and recent water clarity data for Crystal Lake, Vilas County — a very oligotrophic lake on which a relatively large data base exists—Baker and Magnuson (1976) found what they believed to be a slight decrease in water clarity.

Our examination of Secchi disc data on Wisconsin lakes showed that some data are available on a great many lakes, but in no case could definite trends be observed. Plots of summer Secchi disc readings for three lakes; Big Green, Green Lake County, Devil's Lake, Sauk County, and Trout, Vilas County) are shown in Figure 84 to illustrate the variability that occurs and some of the difficulties encountered in assessing trends. The Devil's Lake data seem to reveal a trend toward poorer water clarity in recent years, but the only two Secchi readings made in 1945 and 1955 indicate clarity was about the same then as in 1977-80. It is possible

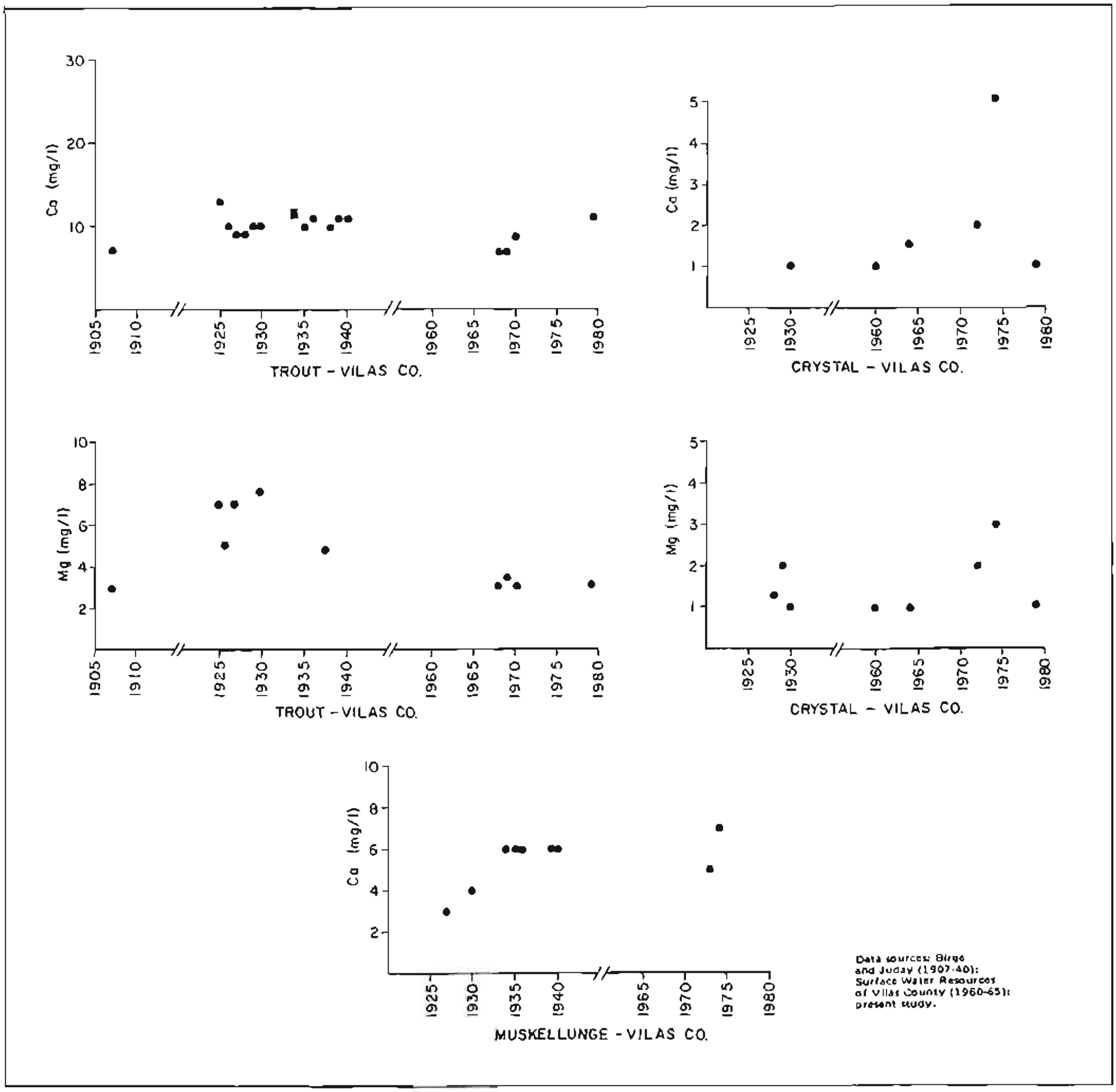


FIGURE 83. *Surface calcium and magnesium concentrations in some Vilas County lakes.*

the current trend could, therefore, be only temporary. For Big Green Lake, the 1980 Secchi reading (3 ft) was the worst ever recorded; however, previous data suggest no change in summertime water clarity between 1900 and 1979. Because only one reading was taken in 1980, it is impossible to determine the significance of it on a long-term basis. Data for Trout Lake show no discernible change in clarity, but long-time gaps when little or no data were collected preclude definite determination of trend.

In summary, only sketchy and generally insufficient data were available for determining historical trends in water quality of Wisconsin lakes; therefore, few trends could be positively identified. In many instances the small amount of data available suggests that no recognizable water quality changes are occurring, but this does not preclude the possibility that in some lakes subtle and long-term changes could be taking place. Even in cases where changes appear to be taking place, no definite cause-and-effect relationships can be established, and discussions of possible reasons for apparent changes can only be speculative.

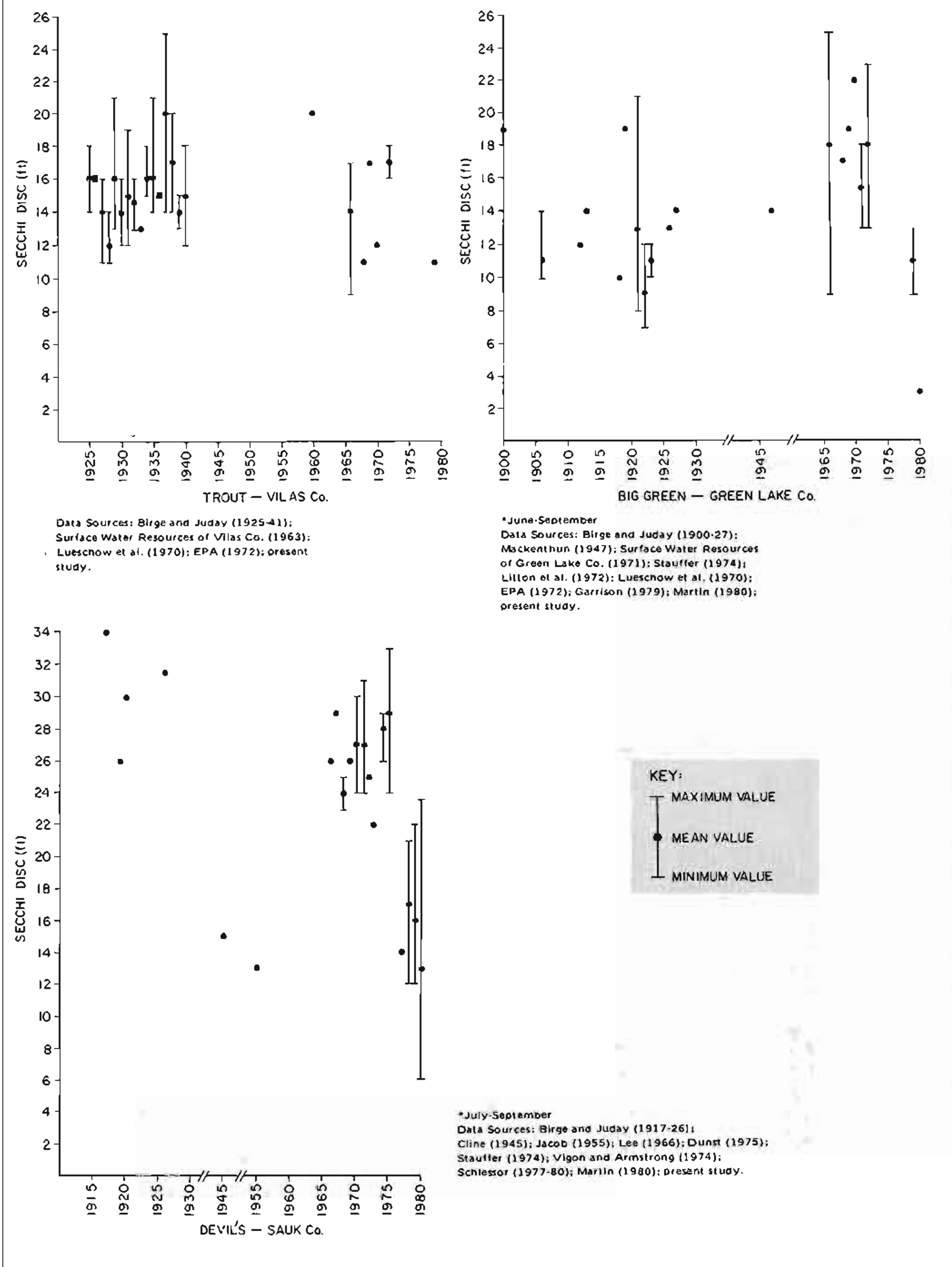
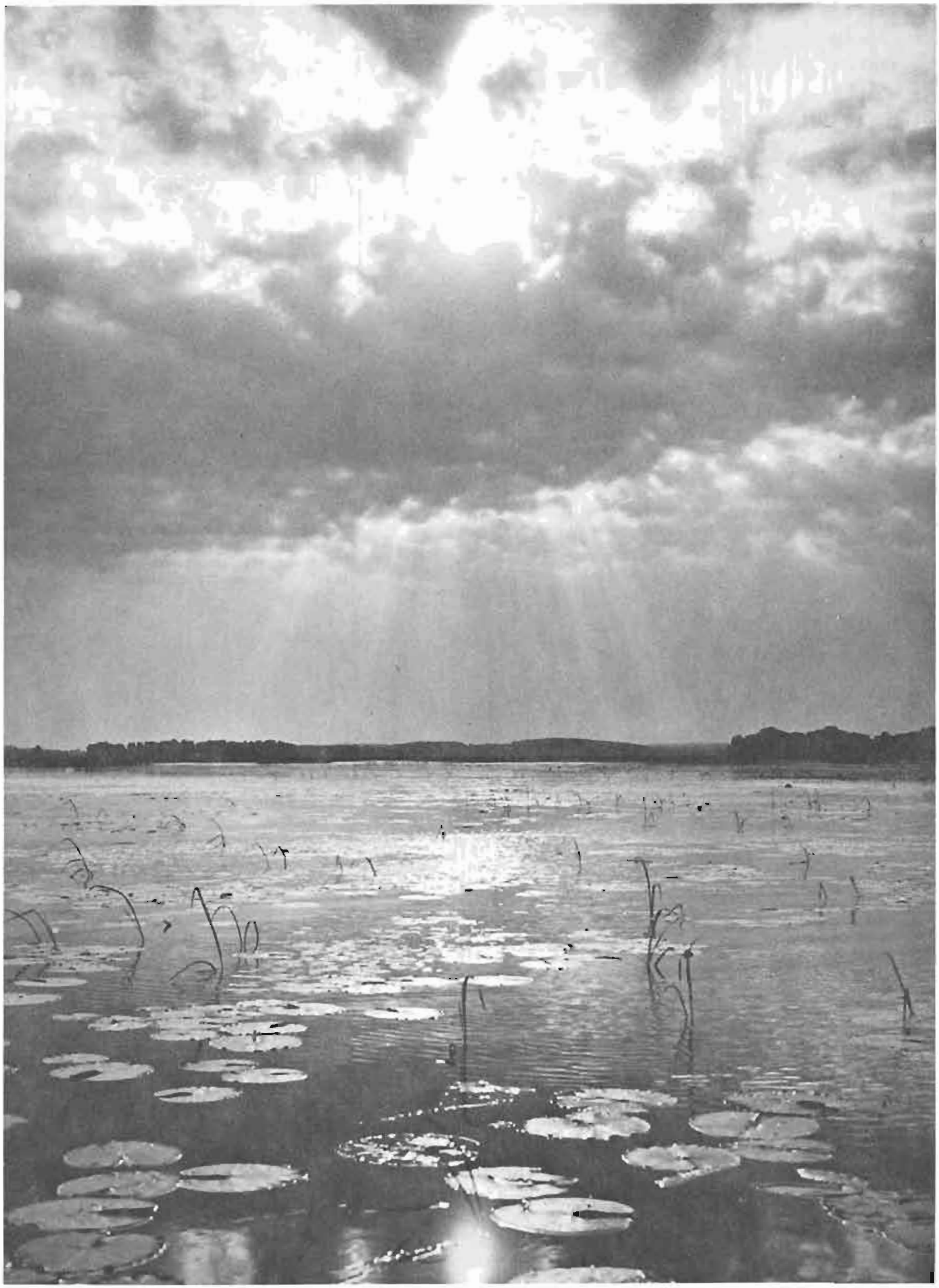


FIGURE 84. *Historical trends in summer water clarity in selected Wisconsin lakes.*

CONCLUDING DISCUSSION



The future water quality of Wisconsin's lakes lies in the wise management of our natural resources. Shoreline and land use management, nonpoint source pollution control, and control of toxic wastes are all critically important elements in protecting Wisconsin's many lakes.

GENERAL

The substantial amount of individual lake data collected and analyzed during this 14-year lake sampling program should be useful as a historical data base for evaluating future trends in water quality for those lakes which were extensively sampled. Lack of a sufficient historical data base has hampered past efforts to determine water quality trends in Wisconsin lakes. We believe the large data set collected through this program may prove to be helpful to lake managers for evaluating future water quality changes or trends.

In addition, the data available on 1,140 lakes should be of value in the process of selecting study lakes, where lakes with particular characteristics and background data are needed.

Statistical data provided in this report could assist investigators in the design and development of lake sampling programs, as described by Dunnette (1980) and Reckhow (1978*a*), thereby economizing on the time and money needed to obtain the desired sampling accuracy. Of particular importance in the design of lake studies is that some parameters are much more variable than others over time and within different lake types; therefore, sampling schedules should be designed accordingly. For example, the phosphorus content of lake waters has been shown to have great importance in assessing water quality status, but it is an extremely variable parameter. Because of this, the accurate assessment of phosphorus means for lakes requires more frequent sampling than would be required for some other water quality indicators.

The great variability in the data within our lake data sets emphasizes the complexity of the interactions occurring between the different chemical, physical and biological characteristics of lakes. Perhaps the greatest area of unknowns lies in the dynamic processes occurring within lakes due to biological activities. Plant-animal relationships are extremely complex, and their impact on lake systems should not be underestimated. Studies have demonstrated that interactions between fish, zooplankton, phytoplankton and macrophytes have profound effects on lake water quality (Brooks and Dodson 1965, Gliwicz 1977, Shapiro 1978, 1979, Carignan and Kalff 1980), and gaining a better insight into these interrelationships could lead to new methods for improving water quality and benefiting fish communities.

LAKE MANAGEMENT

Analysis of the Wisconsin lake data has given support to some previously advanced theories and results of earlier investigations regarding limnological interactions in lakes. Some of these findings appear to have significance for efforts to preserve or enhance the water quality of some lakes or impoundments in the state. The relationship of stratification to summer water quality seems to have significance for designing new impoundments or modifying existing ones, and for planning dredging projects for certain lakes. Impoundments and lake rehabilitation projects involving dredging could be designed to create sufficient water depths to ensure that stratification will occur. It appears to be beneficial to design impoundment or dredging projects to create the largest possible area of hypolimnion. In Wisconsin lakes, depths needed to ensure that thermal stratification will occur may generally be described as:

$$\frac{\text{maximum depth (m)} - 0.1}{\text{Log}_{10}\ \text{lake area (ha)}} > 3.8$$

(Lathrop and Lillie 1980).

Other important factors which influence water quality also have to be considered, including the sedimentation rates, drainage basin sediment export rate, lake flushing time, location of inflows in relation to outlet (if any), and composition and slope of the bottom in the littoral zone.

Based on previous reports and data from this study, it appears that reducing the drainage basin: lake area ratio, or increasing the retention time should be beneficial to a lake system. This conforms to some eutrophication models which predict an improvement in water quality if mean depth is increased (retention time would theoretically also increase). Diversion of inflow around a lake reduces the effective size of the watershed and nutrient loading and therefore should generally result in long-term improvement in water quality, but can also lengthen the retention time. However, this generalization cannot be universally applied to all lakes and impoundments due to other influencing factors (Hutchins 1977, Uttormark and Hutchins 1978). We found that many impoundments and unstratified lakes had lower algal biomass (chlorophyll *a* levels) than would be expected based on their nutrient concentrations. In some cases this low response may be attributed to high flushing rates and turbidities which prohibit complete utilization or conversion of nutrients into biomass. Cellular washout is thought to be an important consideration in impoundments or drainage lakes with very high flushing rates (Uttormark and Hutchins 1978). If water inflow to these lakes is reduced, the retention time will increase and the external nutrient loading will decrease but resulting water quality may not improve immediately. Because the retention time is increased, the lake's sensitivity or response rate to changes in nutrient loading is decreased. Therefore, water quality changes will be dependent on the lake's morphometry (primarily mean depth), the percent area of the lake's sediments exposed to epilimnetic recycling, and the proportion of the nutrient loading attributable to in-lake nutrient recycling vs loading from external sources. Also, a decrease in hydraulic loading (increase in retention time) may increase the phosphorus retention coefficient (Kirchner and Dillon 1975); therefore, the desirability of diversion as a lake restoration or protection technique appears to be highly dependent on individual lake characteristics. A possible example of the beneficial effects of a naturally occurring diversion may be Pike Lake, Washington County, which has better water quality than expected based on inflow nutrient loading, due to the apparent short-circuiting of flow from inlet to nearby outlet (Mace pers. comm.). Additional examples may be represented by the lower Madison lakes (Lathrop and Johnson 1979) and Mirror and Shadow lakes of Waupaca County (Garrison and Knauer 1982).

NITROGEN LIMITATION

Our study suggests that a small but important number of lakes in the state may be nitrogen limited rather than phosphorus limited. The random data set showed only a relatively small percentage (8%) of the state's lakes might be nitrogen limited. However, the quarterly data set, which had a greater proportion of highly eutrophic lakes, had a greater number of possible nitrogen-limited lakes (16%). This may be of significance in lake management efforts to control bluegreen algae blooms, which are often associated with low N:P ratios. The addition of inorganic nitrogen to low N:P ratio lakes has been suggested as a means to shift the phytoplankton community from bluegreen bloom-forming species to green algae which are more readily utilized as food by zooplankton. Such a shift should theoretically be accompanied by an improvement in perceived water quality as a result of better trophic utilization of available nutrients. The reduction of phosphorus, without controlling nitrogen, could result in the same benefits.

TROPHIC CLASSIFICATION

"Trophic classification" of lakes is currently a popular exercise that has been encouraged by federal requirements (amendments to Section 314(b) of the Clean Water Act) for states to categorize their lakes on the basis of "trophic condition". Unfortunately, "trophic condition" is very difficult to define, and the federal definition (P. L. 35.1605-6) — "A relative description of a lake's biological productivity based on the availability of plant nutrients" — is rather vague. Therefore, many different interpretations have been made and many classification systems have evolved. All of the best-known classification systems that have been devised are logically based and useful for purposes intended, but they all have limitations that can lead to misconceptions if these limitations are not recognized. Recent attempts at trophic classification of Wisconsin lakes have relied most heavily on Carlson's (1977) Trophic State Index (TSI) (Sloey and Spangler 1978, Martin et. al. 1983). Carlson's index and subsequent numerical categorization is based on any one or combinations of three variables: (1) water clarity, (2) total phosphorus and (3) chlorophyll *a* concentration. Thus, any lake may have as many as 4 (including composite) TSI's. As a result, lakes so classified may demonstrate a wide range in trophic state index numbers. Seasonal differences or trends in TSI's or disparities in TSI values (same date) may be of practical significance or have implications to lake management. Such changes or discrepancies may be indicative that particular management problems or situations (i.e., high suspended solids) exist and that management approaches must consider the consequences of a particular action.

However, a basic drawback to this form of trophic classification is the fact that the resultant index number and relative placement of an individual lake in or amongst a group of lakes may or may not be at all related to the lake's "true" trophic state in that it ignores other forms of biological production.

Porcella et al. (1980) developed the Lake Evaluation Index (LEI) in order to monitor the limnological changes associated with lake restoration projects throughout the United States. The LEI incorporates a measurement of macrophytes (percent of available lake area covered) with other standard trophic variables (pelagic zone) and thus overcomes the major weaknesses of the TSI. This system is quite comprehensive but it was not intended primarily as a trophic state index (Porcella et al. 1980). While the effects of restoration techniques can be evaluated on the basis of the single whole lake LEI value, the assessment of shifts in the individual variables appears to present the most information to the lake manager. All existing indices which attempt to package data representing a number of variables into an absolute one-dimensional scale (i.e., range 1-100) representative of overall water quality or trophic status have one fundamental flaw. That is, while water quality or trophic status may be expressed along a one-dimensional axis, the parameters used in many of the indices often conflict with one another (i.e., high percent macrophytes and low chlorophyll, or vice versa; low or high total phosphorus associated with high color, poor Secchi disc visibility, etc.). The existing data available for Wisconsin lakes suggest perhaps that comprehensive evaluations lead to oversimplification of a more complex issue. A more desirable approach if one wishes to make comparisons of lakes on the basis of several key parameters may be the development of models which would permit evaluation and comparisons of lakes along three- or four-dimensional axes.

An additional concern regarding trophic assessment is that of secondary production. The term "trophic", as used in the context of lake management, refers to "nutrition" or "growth", and when combined with "state" or "status", referring to "condition", means "condition of nutrition or growth". Nutrition may refer to plant and/or animal incorporation or utilization of food substances. Therefore, assessment of the "true" trophic state of a lake would have to include its total plant and animal life. This approach differs from current assessments of "trophic condition" which for the most part appear to be considering algal production only. The present approach and its dependency upon the standing crop of primary producers seems questionable in light of knowledge concerning the impact of ecological interactions (e.g., herbivore cropping, zooplankton-phytoplankton-fish relationships, etc.) and possibly toxic compounds (e.g., herbicides, heavy metals, etc.) upon primary producers. A problem with basing trophic status on the "sum" of plant and animal life is whether it should be based on rate of production (growth rates), net production (standing crop) or gross production (total annual or seasonal production).

Evaluation of a lake's "true" trophic state thus becomes a very complex issue; algal, macrophytic, invertebrate and vertebrate production or productivity must all be measured and given some value which, theoretically at least, would be more indicative of the lake's nutritive condition. To the best of our knowledge this has not yet been done, although Lindeman (1942) initiated such research attempts using data from, among others, Lake Mendota. Such analyses would be very costly and time consuming, thus restricting the number of lakes that could be evaluated.

This is why more practical measures such as Carlson's (1977) TSI or such multi-parameter evaluations as Uttormark and Wall's (1975*a*) Lake Condition Index or the LEI of Porcella et al. (1980) have become so popular with limnologists who are attempting to classify lakes.

The application of these indices in classifying Wisconsin lakes according to their perceived trophic status is valid for general purposes. However, we believe that serious consideration should be given to clarification of the present policies concerning lake trophic classification including: (1) redefining "trophic state" in relationship to the objectives in mind, (2) clearly stating the objectives for classification, and (3) classification of lakes based on parameters shown to be directly related to the definition of "trophic condition". In turn, impetus should be given to the development of practical, cost-effective, alternative methods of classification which would allow for the categorization of Wisconsin lakes according to total primary and secondary production. (Justification: some lakes exhibiting high trophic classification index numbers (eutrophic) also possess some of the state's most valuable fish resources.) Practicality must also be kept in mind in devising new classification schemes.

We can continue for theoretical purposes to classify lakes in an attempt to discover or describe groupings in which they or certain of their characteristics fall, but unless we can develop quantitative indices, our results will become philosophical exercises unavailable to the wider world.

WATER QUALITY

This study revealed that in lakes with low color and low turbidity levels, the water generally appeared pleasant to observers ("blue" or "clear") when chlorophyll *a* levels were less than 10 µg/l. Thus, preventing chlorophyll *a* levels from reaching 10 µg/l, or reducing them below that level in lakes where they already exceed it, could be regarded as a possible management goal for some lakes. In the absence of color and turbidity, lakes with chlorophyll *a* content of less than 10 µg/l should be aesthetically pleasing and excellent for recreational purposes.

If we accept that a 10 µg/l chlorophyll *a* concentration is or would be a worthwhile in-lake management objective for all lakes (without considering the important role of aquatic macrophytes or the differences in chlorophyll content of the various algae), the question could be raised as to how that goal could be achieved. Presently, most management philosophies, policies and guidelines are based primarily on phosphorus management. When either point or non-point source pollution control problems are addressed, phosphorus is generally the key constituent upon which regulations or control measures are proposed or based. Reduction of the input of phosphorus to any lake regardless of its existing condition, basin morphometry or biota should be beneficial in the long run; but, because of the slow response of some lakes to this reduction in phosphorus inputs and the natural variability in water quality indicators, immediate improvements are often difficult to document and may not be perceivable by the public. However, just because an immediate improvement is not observed does not mean that existing controls are ineffective or should be discontinued, nor does it suggest that further controls are necessary. If phosphorus inputs are controlled, most lakes, according to accepted phosphorus modeling theories, should eventually reach a new equilibrium at a lower trophic level. Based on our study and other lake sampling programs, it is readily apparent that reduction of phosphorus inputs to some of Wisconsin's largest and most important lakes will not bring observable improvement in eutrophic symptoms for many years due to internal phosphorus recycling and other factors. We believe it is important for the public to fully understand this situation so that realistic lake management plans can be formulated.

There are certain practical limitations (both economical and technological) to the application of a phosphorus management policy to Wisconsin's diverse lakes. Bouldin et al. (1977) present an excellent discussion of the alternatives available for the management of water quality in lakes (New York State) using various control strategies for phosphorus. The authors demonstrate that two major sets of policies are available: (1) a varied set of policies which would attempt to establish uniform water quality in all lakes, and (2) a uniform policy or regulation which would be applied to all inputs to lakes and result in different water quality conditions in each lake.

The first option, a varied set of policies to obtain uniform water quality, is economically infeasible and/or technologically impractical for all lakes at the present time. Even if a uniform phosphorus standard were established and were attainable, there is no guarantee that chlorophyll *a* concentrations or water clarity will respond to the reduced phosphorus levels or that the desired improvements in water quality as perceived by lake users will be attained. (It should be made clear that the apparent water quality index parameter values provided in Table 21 are only representative of average conditions based on all random Wisconsin lake data and in no way should be construed to represent definitive lake standards.)

The second option, a uniformly applied policy, is not necessarily justifiable in all cases and may even be detrimental in some cases (i.e., decreased fish production as a result of too few nutrients). It is beyond the scope (or expertise of the authors) of this report to discuss all the pros and cons concerning application of a statewide uniform phosphorus-loading management strategy. However, depending upon the loading changes which occur, such a policy may have limited impact on perceived water quality conditions (see Gakstatter et al. 1978).

Bouldin et al. (1977) suggest that appropriate water quality can be achieved at lowest cost by selective use of policies. The DNR Lake Standards Task Force (Lathrop et al. 1981) reached a similar consensus when suggesting that different sets of water quality objectives be established for Wisconsin's lakes based on existing lake water quality conditions and the economic and technological feasibility of protecting lakes with already satisfactory water quality conditions or improving others wherever possible.

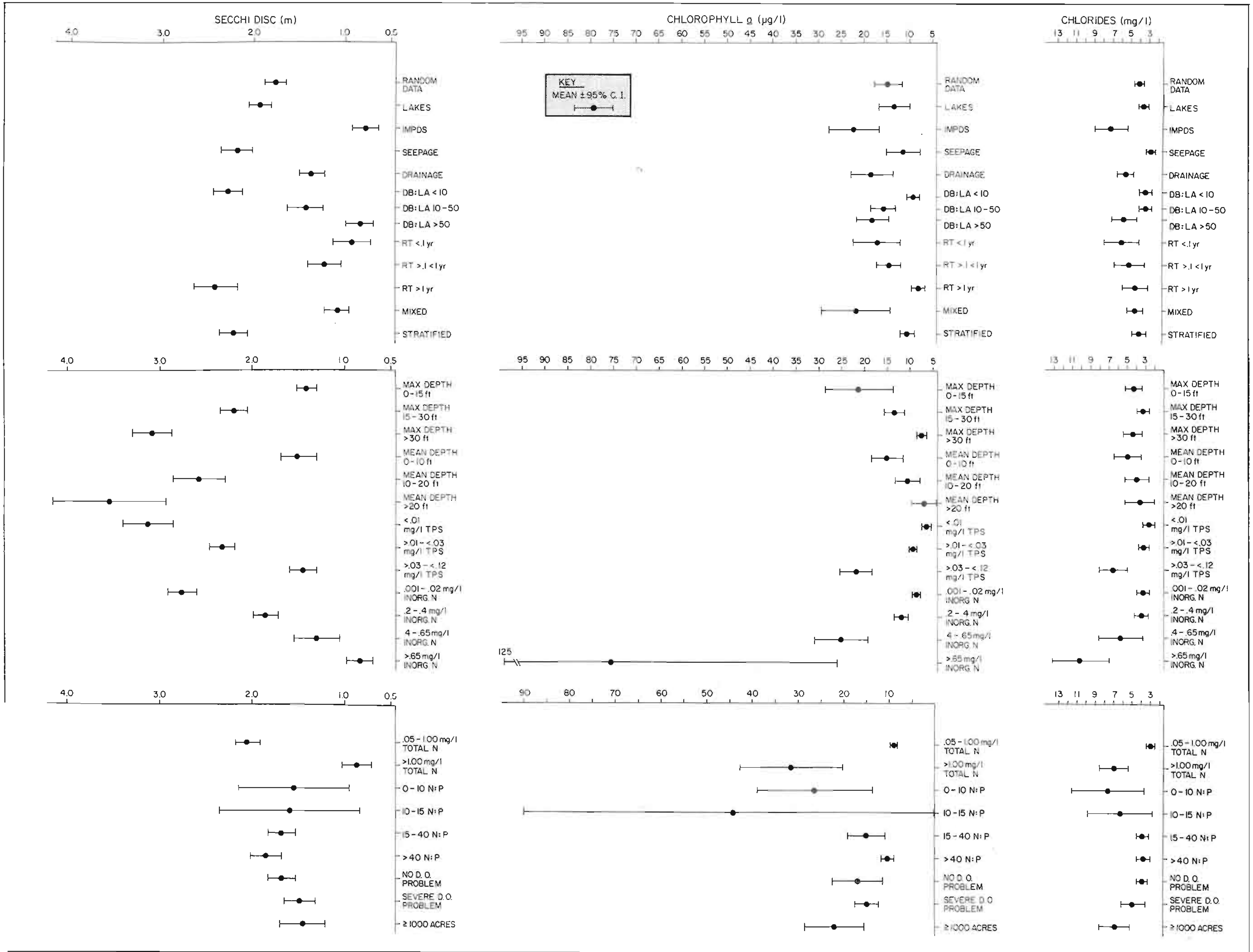
Based on our studies and those of others, we concur with the aforementioned suggestions and further recommend that any form of phosphorus control program for Wisconsin lakes — land use management, detergent phosphorus ban, etc. — must take into account the various limnological characteristics and associated dynamic processes of the lakes involved. Because of the diverse nature of Wisconsin lakes and the complexities involved in attainment of specific water quality objectives, development of sound management strategies must be a cooperative effort between many different interested and knowledgeable groups of professionals, lawmakers and members of the public. We endorse this approach to lake management and hope that the data provided in this report will serve as an aid in the establishment of rational water quality objectives and in the further development of management strategies for Wisconsin lakes.

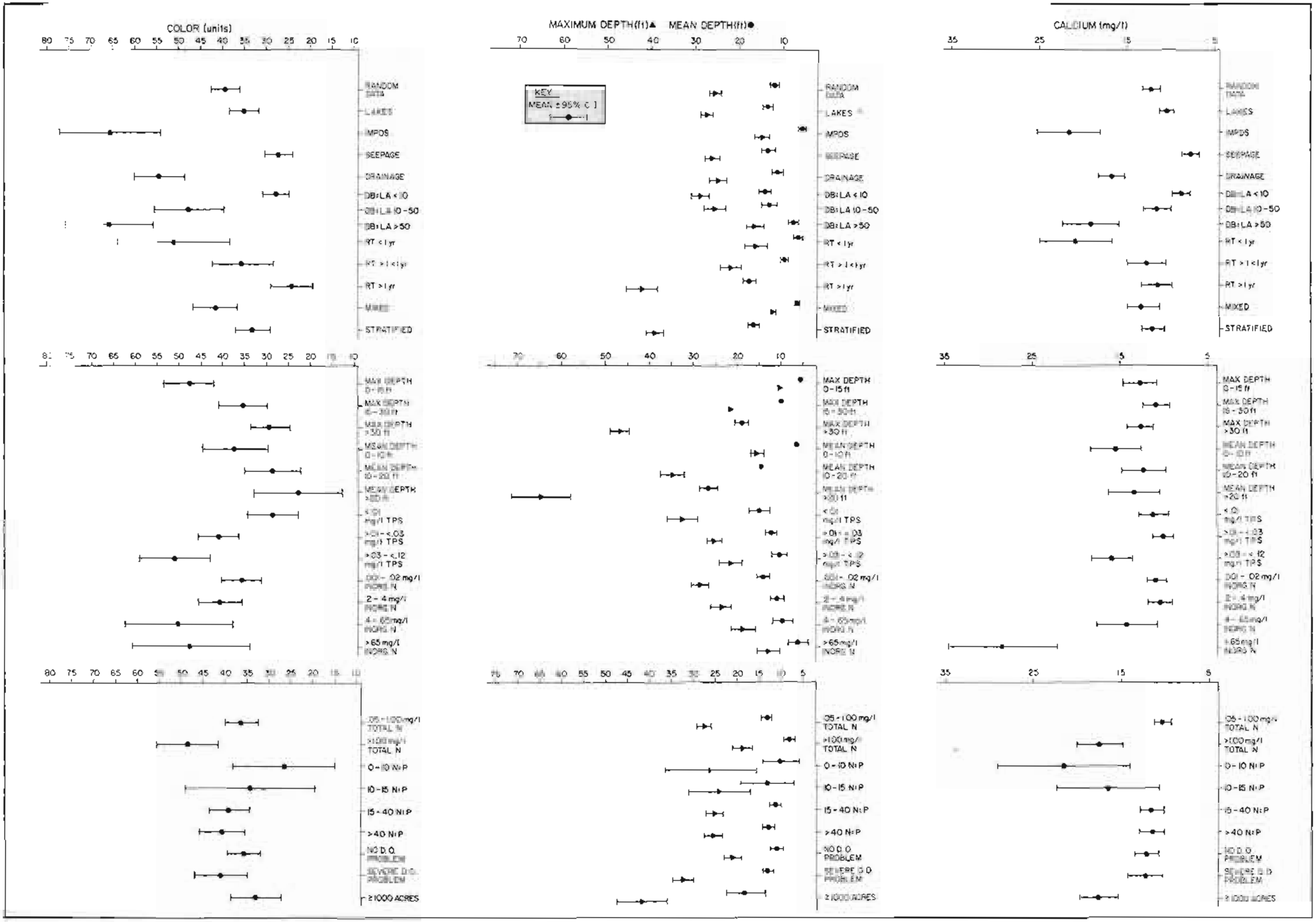
FURTHER CONSIDERATIONS REGARDING MACROPHYTES

Nearly all the currently used lake water quality and trophic classification indices are based primarily on algal standing crop and its effect on water clarity. In our random survey we found that, based on chlorophyll *a* analyses (> 10 μg/l), 65% of the lakes sampled had either an algal bloom in progress or the potential was there for development of algal blooms of some degree. Thus, it appears that a large number of Wisconsin's lakes have algal populations high enough during the summer, at least occasionally, to impart color and/or reduce water clarity to a point that is aesthetically unpleasant. Macrophytes, on the other hand, affect aesthetic and/or recreational quality of only a small percentage of Wisconsin lakes; a DNR Bureau of Fish Management survey showed only 5% of Wisconsin's lakes had a macrophyte "problem", and our survey also indicated macrophyte growth in most lakes was not of sufficient magnitude to cause use problems. Based on these data, it seems that algal growth may be a more widespread "problem" in this state than macrophyte growth. Water quality classification (as opposed to trophic state assessment) on the basis of algal growth (chlorophyll *a* concentration) and the use of water clarity-chlorophyll relationships appears to be a logical approach for many lakes, provided enough samples are collected on individual lakes to determine variability. But even though algal growth may negatively influence water quality conditions in a greater number of lakes than macrophytes, aquatic plants probably have greater impact on lake use and are a bigger problem to lake managers, since problem-causing plant growths are found in some of the state's largest and most heavily used lakes and impoundments, particularly in the southern part of the state. The importance of the macrophyte problem is attested to by the fact that of the 130 lake districts formed in Wisconsin for the purpose of lake restoration or protection, 52% sought aid to combat macrophyte problems while only 25% had primarily algae problems.

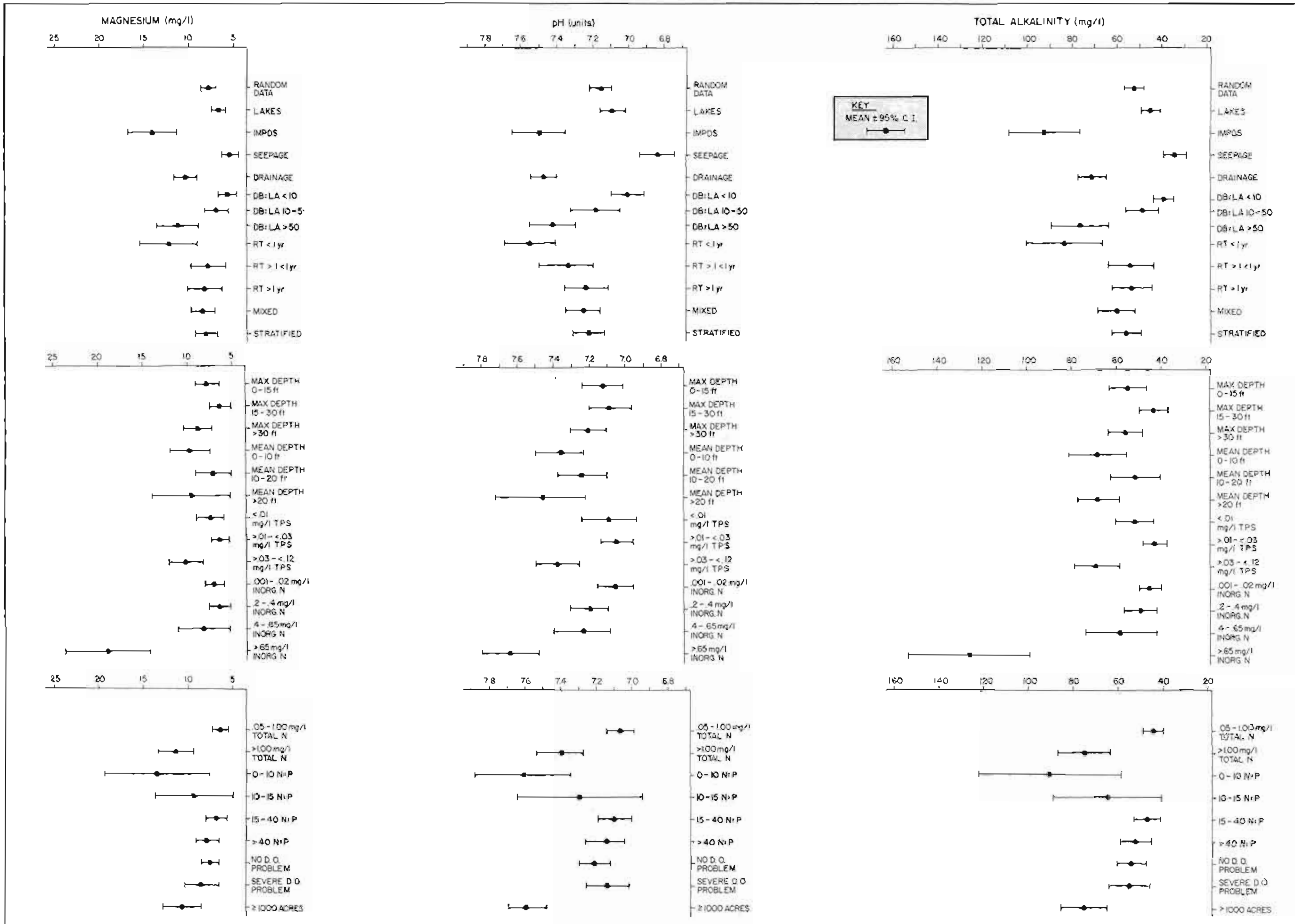
The role macrophytes play in the aquatic ecosystem is not well understood, but in many lakes they are highly desirable aesthetically and for fish and wildlife production. It is only where macrophytes interfere with recreation or become unsightly that problems develop. Such conditions warrant sound and wise management of our natural resources based on well-designed research and exploratory management applications. The management of aquatic macrophytes in individual lakes should be accomplished in such a manner as to provide the maximum diversity of habitat types for fisheries and wildlife while enhancing recreational opportunities for the public where such objectives are indeed compatible.

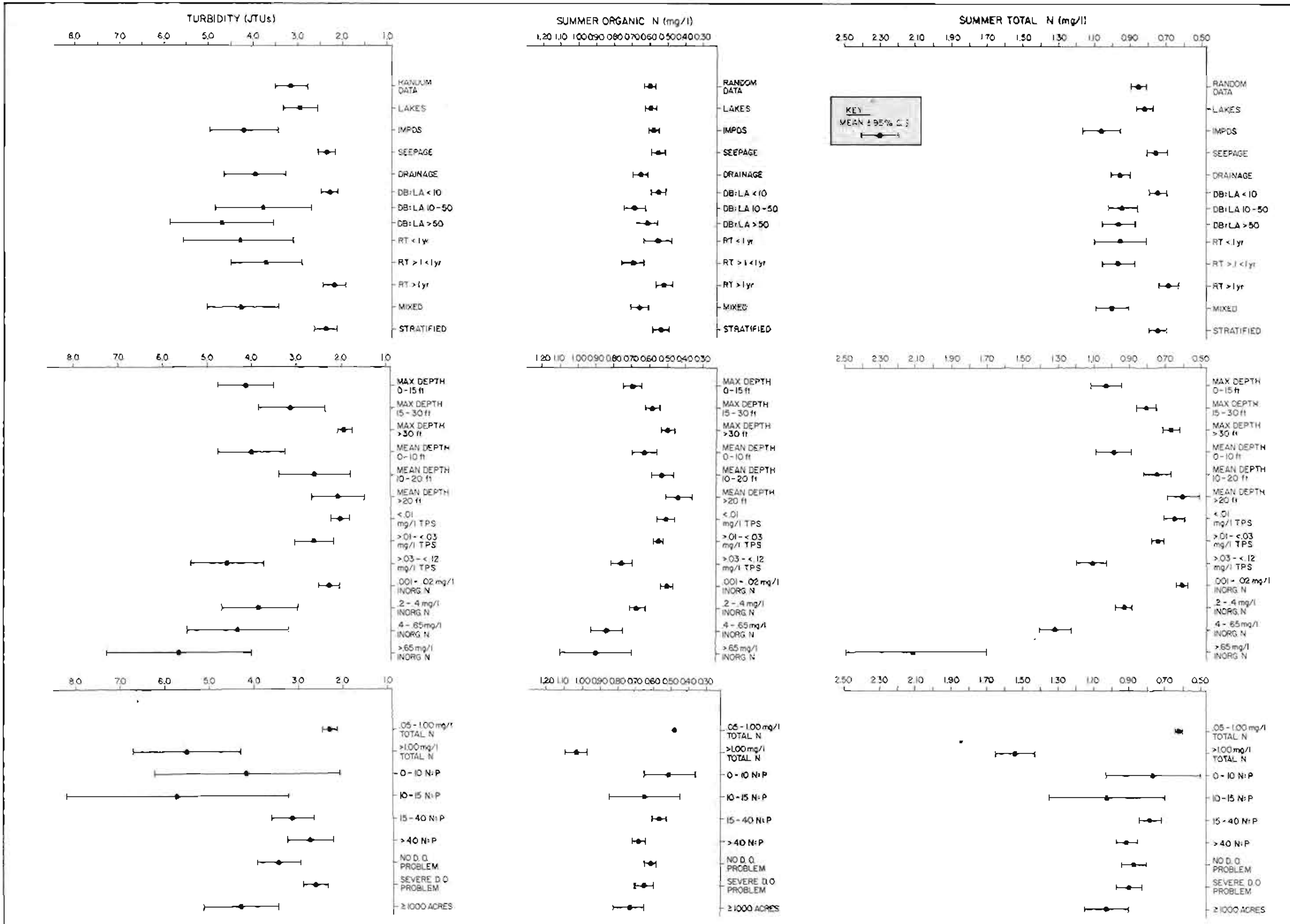
APPENDIX A. Comparison of Lake Characteristics Within Selected Categories of the Random Data Set.

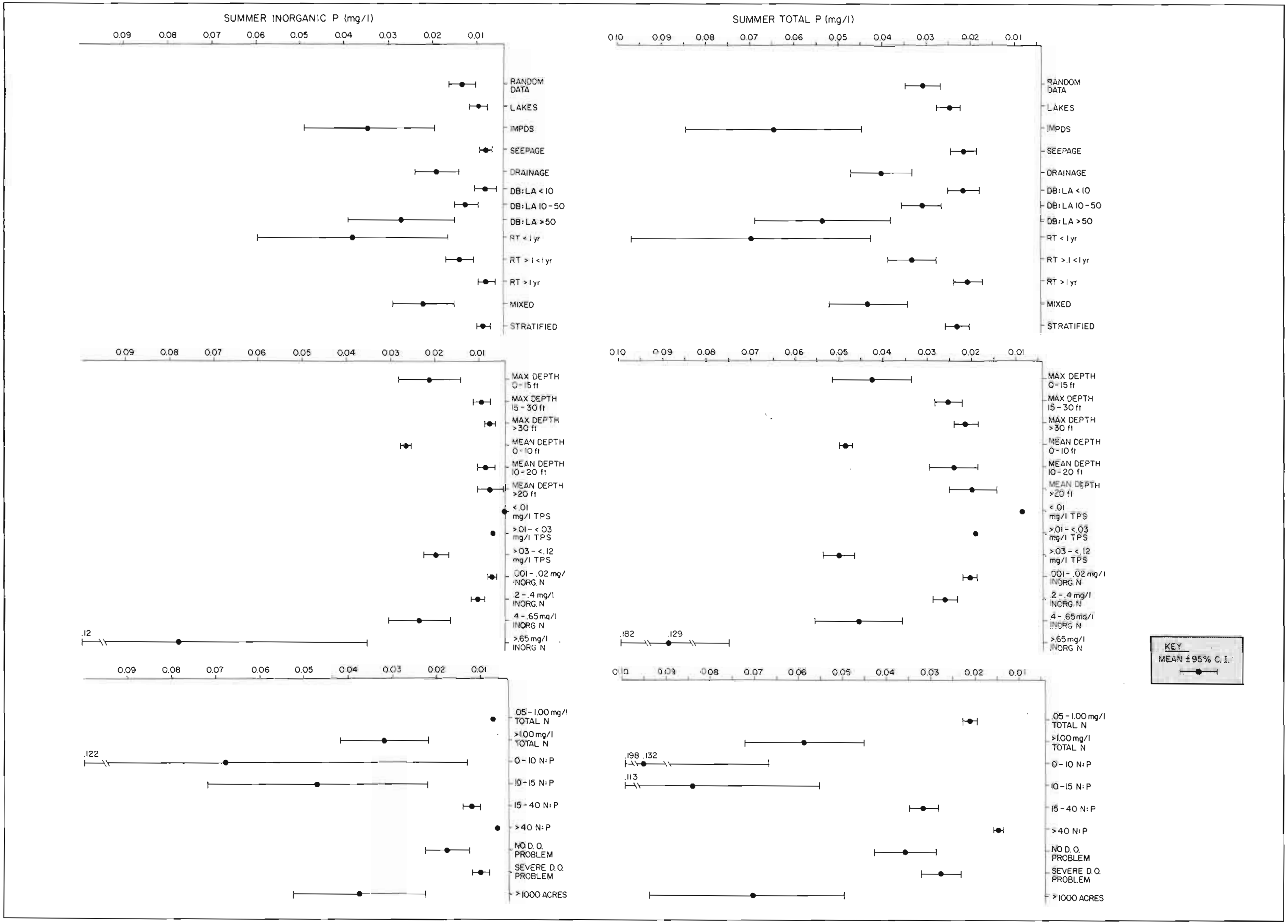




COLOR (units)
MAXIMUM DEPTH(ft)▲ MEAN DEPTH(ft)●
CALCIUM (mg/l)
KEY
MEAN ± 95% C.I.
RANDOM DATA
LAKES
IMPDS
SEEPAGE
DRAINAGE
DB:LA < 10
DB:LA 10-50
DB:LA >50
RT < 1 yr
RT > 1 < 1 yr
RT > 1 yr
MIXED
STRATIFIED
MAX DEPTH 0-15 ft
MAX DEPTH 15-30 ft
MAX DEPTH >30 ft
MEAN DEPTH 0-10 ft
MEAN DEPTH 10-20 ft
MEAN DEPTH >20 ft
< .01 mg/l TPS
>.01 - <.03 mg/l TPS
>.03 - <.12 mg/l TPS
.001 - .02 mg/l INORG N
.2 - .4 mg/l INORG N
.4 - .65 mg/l INORG N
>.65 mg/l INORG N
.05 - 1.00 mg/l TOTAL N
>1.00 mg/l TOTAL N
0 - 10 N:P
10 - 15 N:P
15 - 40 N:P
>40 N:P
NO D.O. PROBLEM
SEVERE D.O. PROBLEM
≥ 1000 ACRES

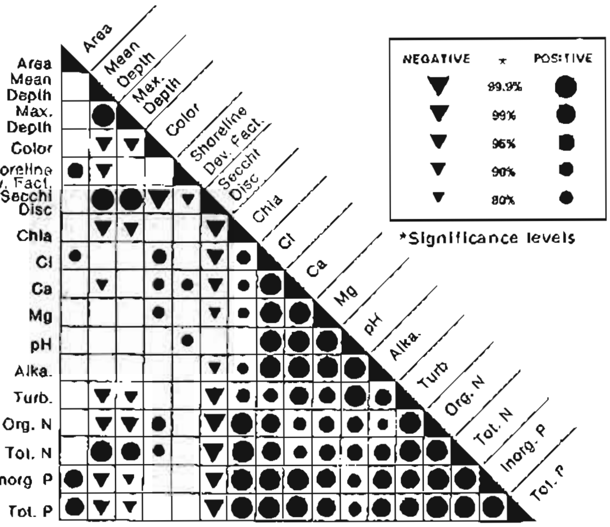

MAGNESIUM (mg/l)
pH (units)
TOTAL ALKALINITY (mg/l)
KEY
MEAN ± 95% C.I.
RANDOM DATA
LAKES
IMPDS
SEEPAGE
DRAINAGE
DB:LA < 10
DB:LA 10-50
DB:LA > 50
RT < 1 yr
RT > 1 < 1 yr
RT > 1 yr
MIXED
STRATIFIED
MAX DEPTH 0-15 ft
MAX DEPTH 15-30 ft
MAX DEPTH > 30 ft
MEAN DEPTH 0-10 ft
MEAN DEPTH 10-20 ft
MEAN DEPTH > 20 ft
< .01 mg/l TPS
> .01 - < .03 mg/l TPS
> .03 - < .12 mg/l TPS
.001 - .02 mg/l INORG. N
.2 - .4 mg/l INORG. N
.4 - .65 mg/l INORG. N
> .65 mg/l INORG. N
.05 - 1.00 mg/l TOTAL N
> 1.00 mg/l TOTAL N
0 - 10 N:P
10 - 15 N:P
15 - 40 N:P
> 40 N:P
NO D.O. PROBLEM
SEVERE D.O. PROBLEM
≥ 1000 ACRES


TURBIDITY (JTUs)
SUMMER ORGANIC N (mg/l)
SUMMER TOTAL N (mg/l)
KEY
MEAN ± 95% C.I.
RANDOM DATA
LAKES
IMPDS
SEEPAGE
DRAINAGE
DB:LA < 10
DB:LA 10-50
DB:LA > 50
RT < 1 yr
RT > 1 < 1 yr
RT > 1 yr
MIXED
STRATIFIED
MAX DEPTH 0-15 ft
MAX DEPTH 15-30 ft
MAX DEPTH >30 ft
MEAN DEPTH 0-10 ft
MEAN DEPTH 10-20 ft
MEAN DEPTH >20 ft
<.01 mg/l TPS
>.01 - <.03 mg/l TPS
>.03 - <.12 mg/l TPS
.001 - .02 mg/l INORG N
.2 - .4 mg/l INORG N
.4 - .65 mg/l INORG N
>.65 mg/l INORG N
.05 - 1.00 mg/l TOTAL N
>1.00 mg/l TOTAL N
0 - 10 N:P
10 - 15 N:P
15 - 40 N:P
>40 N:P
NO D.O. PROBLEM
SEVERE D.O. PROBLEM
≥ 1000 ACRES

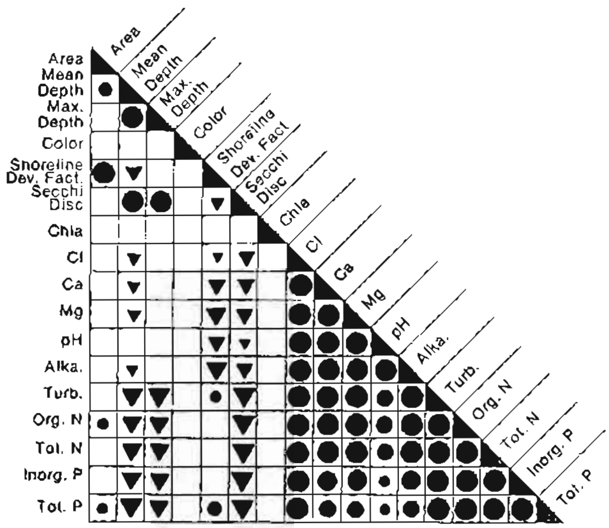

SUMMER INORGANIC P (mg/l)
0.09 0.08 0.07 0.06 0.05 0.04 0.03 0.02 0.01
SUMMER TOTAL P (mg/l)
0.10 0.09 0.08 0.07 0.06 0.05 0.04 0.03 0.02 0.01
RANDOM DATA
LAKES
IMPDS
SEEPAGE
DRAINAGE
DB:LA < 10
DB:LA 10 - 50
DB:LA > 50
RT < 1 yr
RT > .1 < 1 yr
RT > 1 yr
MIXED
STRATIFIED
MAX DEPTH 0 - 15 ft
MAX DEPTH 15 - 30 ft
MAX DEPTH > 30 ft
MEAN DEPTH 0 - 10 ft
MEAN DEPTH 10 - 20 ft
MEAN DEPTH > 20 ft
< .01 mg/l TPS
> .01 - < .03 mg/l TPS
> .03 - < .12 mg/l TPS
.001 - .02 mg/l INORG. N
.2 - .4 mg/l INORG. N
.4 - .65 mg/l INORG. N
> .65 mg/l INORG N
.12
.182 .129
.05 - 1.00 mg/l TOTAL N
> 1.00 mg/l TOTAL N
0 - 10 N:P
10 - 15 N:P
15 - 40 N:P
> 40 N:P
NO D.O. PROBLEM
SEVERE D.O. PROBLEM
> 1000 ACRES
.122
.198 .132
.113
KEY
MEAN ± 95% C.I.

APPENDIX B. Correlation Matrixes Showing Strongest Relationships Between Parameters Based on Different Subsets of Lakes.

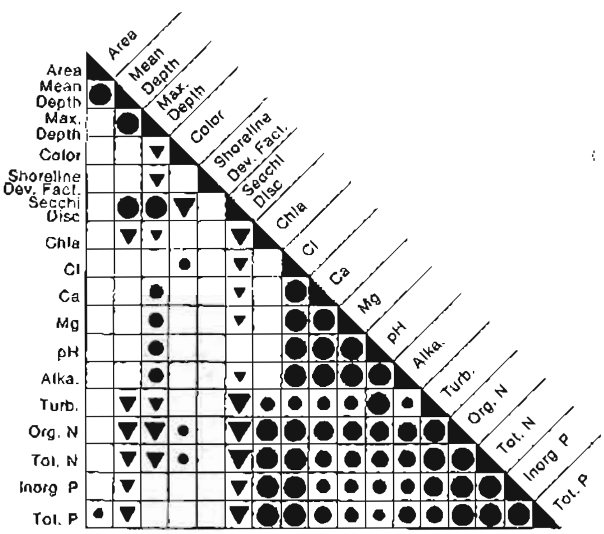
RANDOM DATA SET



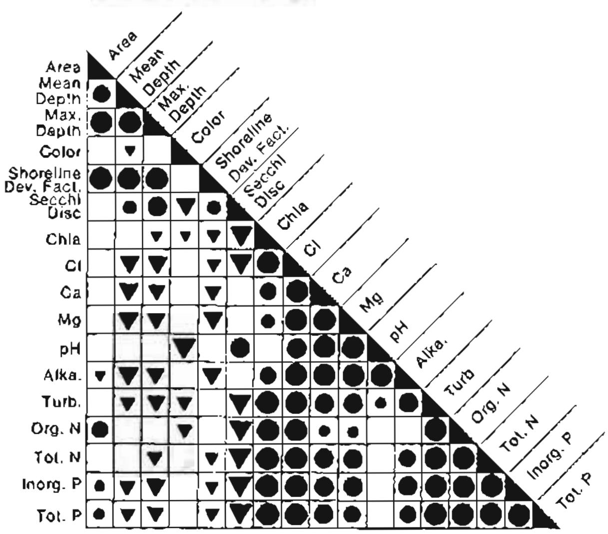
QUARTERLY DATA SET

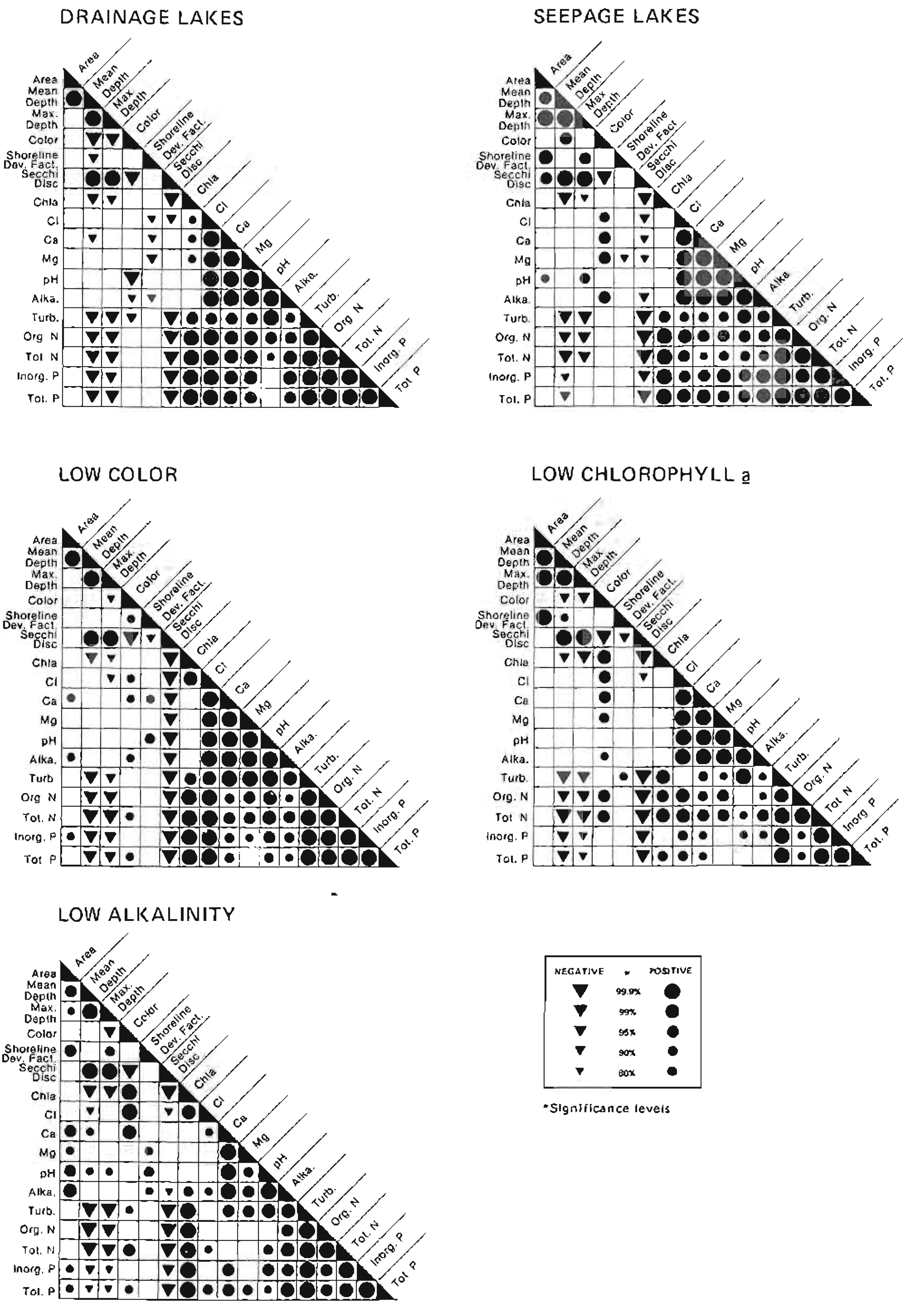


NATURAL LAKES

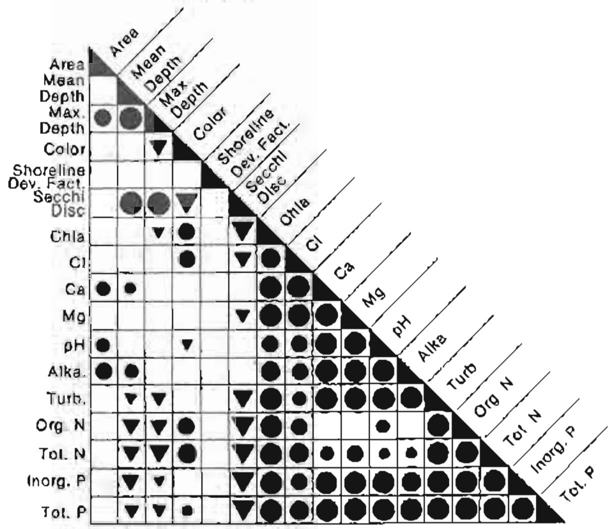


IMPOUNDMENTS

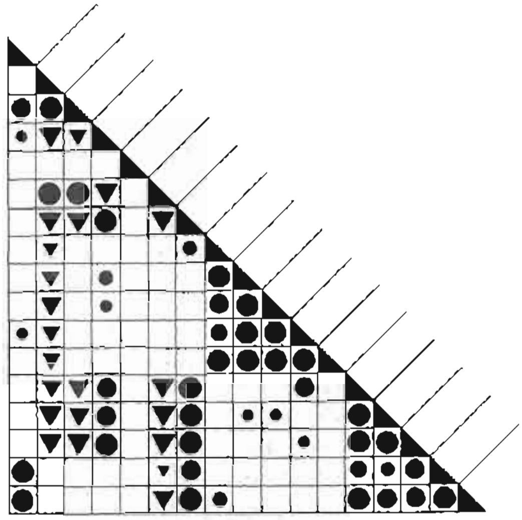



DRAINAGE LAKES
SEEPAGE LAKES
LOW COLOR
LOW CHLOROPHYLL a
LOW ALKALINITY
Area
Mean Depth
Max. Depth
Color
Shoreline Dev. Fact.
Secchi Disc
Chla
Cl
Ca
Mg
pH
Alka.
Turb.
Org. N
Tot. N
Inorg. P
Tot. P
NEGATIVE
*
POSITIVE
99.9%
99%
95%
90%
80%
*Significance levels

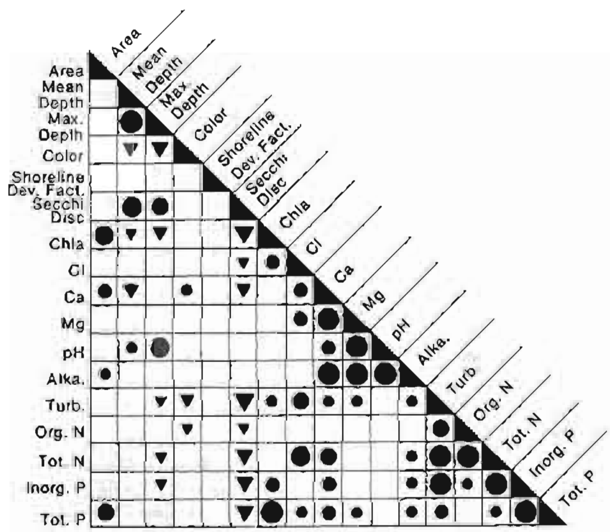
NORTHWEST



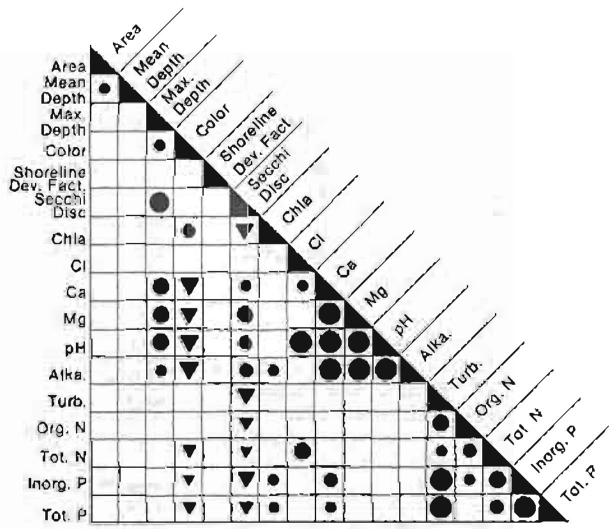
NORTHEAST



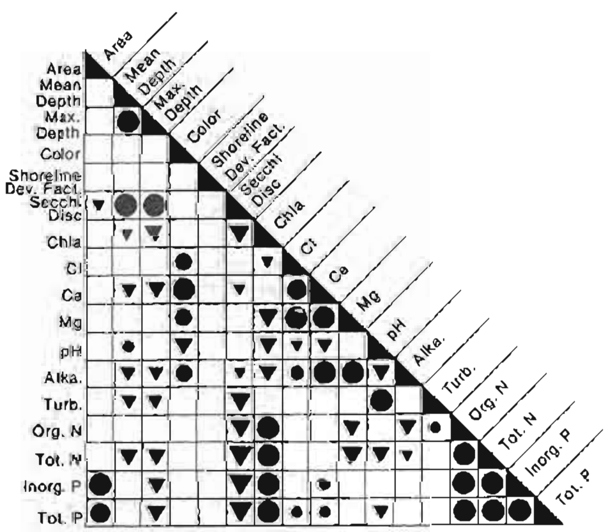
CENTRAL

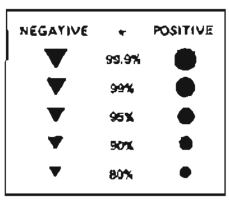


SOUTHWEST



SOUTHEAST





*Significance levels

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